The response of bird communities to forest degradation and eucalyptus plantations in Mount Kenya

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A thesis submitted in partial fulfilment of the requirements of the University of Brighton for the degree of Doctor of Philosophy

February 2023

Abstract

Anthropogenic disturbance is one of the leading causes of biodiversity change in tropical African forests, coupled with high rate of adoption of exotic plantations near natural forests. This is compounded by contemporary forest conservation approaches adopted, such as participatory forest management (PFM), that legally allows local community to use forest resources, leading to local disturbance within the forests. Increase in exotic plantation in Afrotropical forests is exacerbated by the need to increase forest cover and to meet high demands for commercial and non-commercial woods, particularly in montane forests. Mount Kenya's montane forest represents an ecologically important protected Man and Biosphere Reserve and an important bird area. However, it is managed under PFM and planted with exotic plantations on cleared, unforested and degraded forest sites. But the changes in forests' habitat structure and characteristics, and how it affects local biodiversity following these local disturbance and plantations remains significant knowledge gaps in Afromontane forests. The aims of this research are to determine the habitat characteristics in undisturbed, disturbed, and eucalyptus plantation forest types, and the impact of these on bird diversity and community compositions.

Birds and habitat characteristics data were collected for a year in a total of 190 systematically placed point counts distributed in forest types across three study sites in eastern, southeastern, and southern Mount Kenya forest. All forest types were characterised by different habitat characteristics. Undisturbed forest had most forest complexity characteristics, with increasing habitat homogeneity from disturbed to eucalyptus plantation. Eucalyptus plantation had exceptional open canopy, tall, and dispersed trees. Forest types significantly influenced combined bird species, Afrotropical highlands biome restricted species (ATHB), frugivores, granivores and nectarivore dietary guilds, and all forest dependency groups except generalists. Habitat characteristics related to complex forests positively predicted forest specialists', frugivores', insectivores', and ATHB's species richness and abundance. It negatively predicted richness and abundance of forest visitors (FV), non-forests birds (NF), granivores, omnivores and nectarivores. Characteristics in eucalyptus plantation predicted positively FV's, NF's, granivores' and omnivores richness and abundance, and negatively ATHB's,

and frugivores' richness and abundance, and influenced their compositions. Although community composition of birds among forest types revealed a general ecological complementarity, there was important contribution of natural forests irrespective of local disturbance. Eucalyptus plantation represented species more associated with surrounding landscape rather than forests. This research has demonstrated the importance of natural forests irrespective of local disturbance, yet exotic plantations contribute minimally to forest birds in Afrotropical montane forests.

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Author's Declaration

I declare that the research contained in this thesis, unless otherwise formally indicated within the text, is the original work of the author. The thesis has not been previously submitted to this or any other university for a degree and does not incorporate any material already submitted for a degree.

Signed:

Dated: 02/07/2022

Chapter 1 – General introduction

1.1 Tropical mountains

Tropical mountains are among the most biodiverse ecosystems on earth, containing species that are highly adapted to narrow niches owing to low seasonal differences and low temperature ranges (e.g., Dimitrov et al., 2012; Brown et al., 2014; Hoorn et al., 2018; Rahbek et al., 2019), yet highly vulnerable to rapid changes resulting from anthropogenic degradations (Morris, 2010; Christmann and Menor, 2021). But the drivers of tropical mountain biodiversity dynamics have remained less understood, particularly with regards to the contemporary anthropogenic-biodiversity interactions, as influenced by increasing human needs (Peters et al., 2019; Wang et al., 2019).

Mountains by nature generally vary in terms of latitude and altitude (Xu et al., 2017), and in the position of the ecosystem belts occurring along their gradient (Figure 1.1).
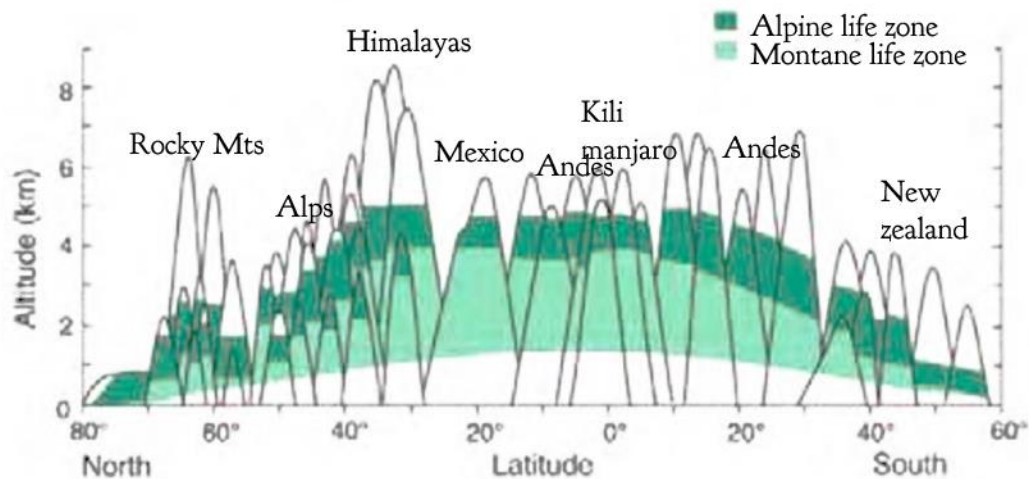
Figure 1.1: Mountain variations in latitude of occurrence and altitude. It also shows relative altitudinal positions of montane and alpine zones across the global latitude in lateral view (Source: adapted from Spehn et al., 2010)

Mountains close to the equator differ in terms of vegetation and animal structure and composition diversity compared to the poles (Molau, 2004). This is because of increased biodiversity from the poles to the equator mainly resulting from increasing temperatures

that raises ecological productivity towards the tropics (Brown, 2014). It is also probably the case based on ecological theories i.e., time theory, local disturbance, spatial heterogeneity theory, among other theories reviewed by Pianka (1966), as well because of evolutionary and historical biogeographic theories (e.g., Wiens et al., 2009). Ecosystem belts in a mountain also differ from lowlands to upper/top part of the mountain (Figure 1.2), which makes species diversity and composition also to differ (Xu et al., 2017). The ecosystem belts within a given mountain differ because of climatic and elevational differences (Spehn et al., 2010), that create conditions upon which different ecosystems form such as montane forests, alpine and nival (Figure 1.2).
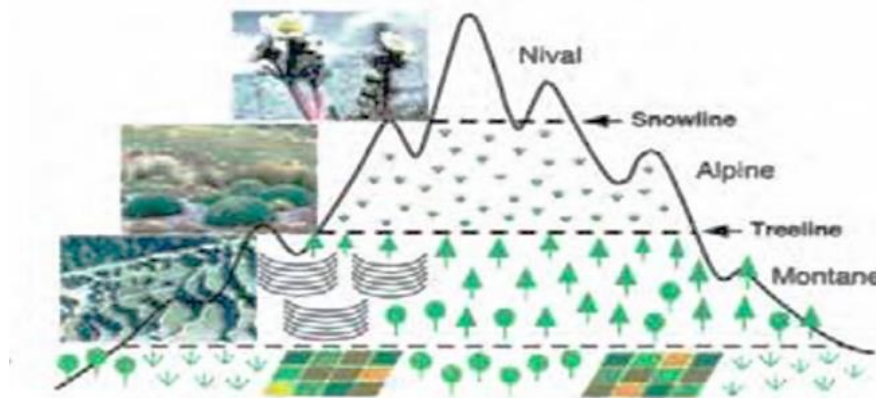
Figure 1.2: Nomenclature of a mountain elevational belts from lowlands to nival. (Source: Spehn et al., 2010**).**

However, montane ecosystems also vary in terms of size and extent, caused mainly by artificially created circumstances such as anthropogenic disturbance of ecosystems within the mountains. This is more pronounced in the tropics (Morris, 2010; Gill and Malamud, 2017) particularly in Afromontane regions where variation of montane ecosystems reflects how human activities have impacted on them (; Feurdean and Astalos, 2005; Borghesio, 2008; Kayombo et al., 2020; Beche et al., 2022). The montane part of the tropical mountain (i.e., tropical montane) covered mainly by forests are especially vulnerable and negatively impacted by anthropogenic activities, degrading it of its former biodiversity integrity (Richter, 2008; Salinas et al., 2021).

1.2 Tropical montane forests

Only 13 % of the area covered by tropical and subtropical forests, are covered by tropical and subtropical montane forests (Salinas et al., 2021), covering about 305 million hectares (FAO and UNEP, 2020). Despite anthropogenic threats (highlighted in section 2.3), tropical montane forests (TMFs), that includes cloud forests, are critical in the provision of ecosystem services, and the regulation of water and the regional climate (Bubb et al., 2004). They are also the richest and most diverse on earth, playing the most important role in biodiversity conservation (Kessler and Kluge, 2008; Richter, 2008). TMFs are where most flora and fauna are ecological specialists with narrow elevational distributions, limited geographic ranges, and small global populations (e.g., Jankowski, et al., 2021).

Exceptional richness in biodiversity in TMFs is due to the following (e.g., Gradstein, et al., 2008; Richter, 2008): (i) typical complex spatial and environmental gradients, (ii) evolutionary time scale, (iii) variety of climatic and micro-climate differentiations, (iv) orographic heterogeneity, geologic and edaphic suitability, and (v) disturbance regimes that potentially generate high diversity of habitats.

1.2.1 Disturbance in Tropical Montane Forests

On disturbance, TMFs are subject to a wide range of disturbance types and regimes, that strongly drive TMF ecology (e.g., Crausbay and Martin, 2016), and impacts biodiversity at ecosystem, community, population, and organism ecological levels (Soh et al., 2019). Some levels of disturbance enhance diversity (e.g., Roxburgh et al., 2004), and others degrade the diversity in TMFs (Cayuela et al., 2006; Alroy, 2017). Yet there is poor research representation especially in African TMFs (e.g., Soh et al., 2019). Available studies on TMFs are more about forest fragmentation, which result from extreme forms of disturbance that causes biodiversity loss (e.g., Newton et al., 2009). But all forms of disturbance are identified to modify habitats particularly closer to montane forest edges (Jankowski et al., 2021). Forest edges are also influenced by neighbouring conditions such as agricultural lands and settlements (Didham et al., 2015; Deikumah et al., 2017; Seifert et al., 2022). Disturbance is potentially able to modify not only forest edges but can extend

to hundreds of meters into intact forest (Didham and Lawton, 1999). For instance, Beche et al (2022) found that anthropogenic disturbance can penetrate at least a kilometer into the forest in Afromontane forest. This may particularly affect forest dependent species particularly those that cannot easily adapt to altered forest vegetation structures and emerging new microclimates resulting from such disturbances (Jankowski et al., 2021; Beche et al., 2022). Local anthropogenic disturbance through logging for timber, fuelwood cutting, browsing by livestock and development of infrastructures, can represent this form of disturbance at the forest edges and within protected forest stands. These forms of disturbance remain most chronic and widespread (Ramírez-Marcial et al., 2001; Aravena et al., 2002; Bleher et al., 2006; Newton et al., 2009), and can change the forest vegetation structural arrangement such as canopy cover, tree heights among other structural characteristics in tropical montane forests, yet are understudied (Soh et al., 2019). In addition, there have been recommendations to consider the multiple scales in forest degradation in tropical montane forests for proper conservation planning (e.g., Soh et al., 2019), especially the fine scale habitat features (e.g., related to forest structural arrangement, i.e., tree size, tree heights, foliage arrangement etc) because they are believed to be better predictors of biodiversity than landscape related features such as isolation, or fragmentation (Cayuela et al., 2006; Banks and Gagic, 2016; Michael et al., 2017). The fine-scale habitat features, and conditions are particularly important in African montane (Afromontane) forests because they are mainly affected by human use (through removal) due to high demands of trees and tree related resources such as timber and fuelwood (Kaburi and Medley, 2011; Ramage et al., 2017) and following ease of access to lowland forest edges (e.g., Leaver et al., 2019; Abiem et al., 2020). This is coupled by effects of winds and drying at forest edges, that also directly affect structure and conditions of vegetation (Jankowski et al., 2021).

Afromontane forests

In the African continent, there exist highlands that are covered to a large extent by Afromontane forests. They typically occur above 1500 m in elevation (White, 1978). These highlands extend from the Arabian Peninsula south along the rift valley to Drakensberg Mountains in the east (Abiem et al., 2020). In western Africa, the

Afromontane forests are represented by forests in the Cameroon volcanic line and the Guinea highlands (Gehrke and Linder, 2014). In east Africa, the 25 Afromontane forests occur as patches of various sizes on continuous and isolated mountain ranges in both Sudan, Kenya, Uganda, and northern Tanzania. The most prominent of these include Mounts Kilimanjaro, Meru, Kenya, and the high elevation areas flanking the Rift Valley. These forests are most highly developed on parts of the mountains that receives the most rainfall (i.e., east/south), with much of the original habitats having been lost where they are not protected (for example at lower altitudes where they are most accessible to humans) (Newmark, 1991; Young, 1996; Lovett and Wasser, 2008).

The Afromontane forests receive rainfall of between 1200 – 2000 mm per annum with wetter months occurring between October- December and March-June while dry seasons occur between January-March and July-October (Young, 1996). The Afromontane soils are complex due to varying climate and altitude, but most areas have volcanic soils, which means they are fertile and highly suitable for agriculture (EAC, UNEP and GRID-Arendal, 2016; Nsengiyumva, 2019). Most areas surrounding the Afromontane forests of east Africa are highly populated with a density of 150-400 persons per square kilometer, which is rapidly increasing in Kenya (Emerton, 1999; Cordeiro et al., 2007; KNBS, 2019), where they have been found to settle right up to protected area boundaries, and in some cases, human encroachment extends within the borders of the reserves themselves (e.g., Gathaara and Leakey, 1999). The forests have therefore been under intense threat from agricultural practices, fire, and grazing (Chapman et al., 2004; Cordeiro et al., 2007).

In east Africa, Afromontane forests contain endemic species, which are more pronounced among small mammals and herpetofauna, with several strictly endemic species of Chameleons (e.g., Cordeiro et al., 2007). Birds show moderate levels of endemism but are reported to be very diverse in Afromontane areas, with species found that have restricted ranges (Stattersfield et al., 1998).

The status of Afromontane forests of east Africa is that they are highly fragmented and remain in isolated blocks (e.g., Aerts et al., 2011), with the main threats being related to the increasing human population. Most east African Afromontane forests are within

National parks and Forest reserves, but habitats are affected by infiltration of human activities, that spill over from surrounding areas into the forests (Asefa et al., 2017; Kayombo et al., 2020). The surrounding areas experience intense human activities which leave modified natural habitats up to the edges of the protected areas, for example, at Kilimanjaro (Newmark, 1991), Kakamaga and Mau forests (Wass, 1995), Mount Kenya (Gathaara and Leakey, 1999) and the Ethiopian montane forest (Asefa et al., 2017). The areas close to Afromontane forests, which are normally at lower altitudes, have been converted to agricultural or other human use (Gathaara and Leakey, 1999) such as tea and coffee plantations, or grazing areas. The human activities that pass into protected Afromontane forests are sometimes ubiquitous and discreet yet potentially damaging to forest habitats and the survival of species. However, knowledge on how local, low-key, and non-obvious disturbances in Afromontane forests affects forest habitats and species diversity are minimal.

In addition, the east African Afromontane forests have been planted with exotic tree plantations (Hulme et al., 2013; Teucher et al., 2020). These trees have been widely planted in response to the need to restore formerly forested land that has been cleared, or to restore parts of the existing degraded areas of the protected natural forests, or just to increase the forest cover (Mansourian and Berrahmouni, 2021). Much of this has been done with the use of industrial monocultures involving particularly *Pinus*, *Eucalyptus*, and *Acacia* species (Lamb et al., 2005). Although these species are preferred for other uses (e.g., timbers, medicines, and foods) rather than for their ecological values, it is not widely known to what extent these species may provide benefits to the new environments.

Mount Kenya forest

One of the key Afromontane forests in east Africa is found in Mount Kenya. Mount Kenya is globally important for biodiversity conservation, having key habitat for endemic, threatened and restricted range species and containing the largest remaining single contiguous forest stand in Kenya (Speck, 1982; Evans and Fishpool, 2001; Bussmann, 2002; Niemella and Pellikka, 2004; KWS, 2010). It is also an internationally recognized UNESCO World Heritage Site and Man and Biosphere Reserve

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(https://whc.unesco.org/en/list/800/) and an important site for recreational and tourism purposes. It is also one of the 62 Important Bird and Biodiversity Areas (IBAs) in Kenya recognized as a site for priority conservation (Bennun and Njoroge, 1999).

Mount Kenya forest management, consisting of KFS and Kenya Wildlife Service (KWS), through their respective management plans (i.e., Mount Kenya Forest Reserve Management Plan (MKFRMP) and Mount Kenya Ecosystem Management Plan (MKEMP), have established management strategies consisting of zonation of Mount Kenya forest to reflect the user intensity of the zones and to ensure that the forest is protected to meet the management needs (KFS, 2010; KWS, 2010). The management by KFS has also initiated plantation forests of mostly exotic species to rehabilitate the degraded areas and to increase forest cover for ecological purposes in Mount Kenya, with the added benefit of commercial trees (KFS, 2010, KWS, 2010).

Local communities depend on the Mount Kenya forest for both socio-cultural, religious, and for wood and non-wood forest products both legally and illegally (Kariuki, 2006; KFS, 2010). The local communities adjacent to Mount Kenya forest, have formed Community Forest Associations (CFAs) through the Participatory Forest Management (PFM) approach, where they participate in forest management activities and are allowed through agreements, to access and use forest to extract some wood and non-wood forest products (Wily, 2002; KFS, 2010; Musyoki et al., 2016). As a result, and due to other national and local economic activities and incentives, the forest has been facing major anthropogenic threats such as encroachment for logging, fuelwood collection, livestock grazing and fires (Gathaara and Leakey, 1999; Kaburi and Medley, 2011). This is in addition to occasional natural threats such as landslides and fires (Nyongesa and Vacik, 2019). Additionally, the large human population (mostly a farming community) surrounding the mountain have created intensive land use change and have more demands for resources from the forest (Emerton, 1999; Gathaara and Leakey, 1999). Tourism related development, and other legally or illegally allowed human activities within the forest have led to disturbance of natural areas and sites (KFS, 2010; KWS, 2010), and are likely to have affected the vegetation structure, and other forest characteristics that offer resources and conditions to various forest species. It is from these pressing issues that

there is need to assess the role of local disturbance in influencing habitat and biodiversity in an Afromontane forest such as the Mount Kenya forest.

Research needs for Mount Kenya forest

The role of local anthropogenic disturbances in Mount Kenya's Afromontane forest have not been widely explored yet. Only a few studies (i.e., Ndegwa, 2014; Kioko et al., 2016; Mahiga et al., 2019) have addressed some aspects of disturbance in secondary forest, distribution, and diversity of birds along an altitudinal gradient, and the general influence of land-use types on forest birds, respectively. There is no knowledge on the impacts of disturbance on vegetation characteristics in different management zones, how it affects birds' diversity, and how bird species and the community respond to it. These remain management constraints highlighted in section 4.3 (a) (i.e., lack of updated information) in the Mount Kenya Forest Reserve Management Plan (MKFRMP) (KFS, 2010). They also constitute some priority research areas identified by Mount Kenya Ecosystem Management Plan (MKEMP) 2010-2020 (KWS, 2010). For example, under its Objective 4, action line number 4.8, on priority management-oriented research, Mount Kenya Ecosystem Management Plan has identified ecological studies such as species-habitat interactions, community-forest interactions, and issues on reforestation as among 13 priority research areas required for the Mount Kenya ecosystem. Specifically, the habitat characteristics and conservation values of different forest types in Mount Kenya, have not yet been investigated. This includes the different characteristics of vegetation structures and usefulness of different vegetation structural characteristics in different forest types, including plantations, for ecological functions and biodiversity conservation. In addition, it is of interest to know how birds relate to different forest types and habitat characteristics. It is interesting to understand how these relationships are affected by local small scale anthropogenic disturbance such as human trails and cut trees, plus other forest structural related habitat characteristics. These are investigated in this study with consideration of different forest dependent birds, dietary groups, and species of conservation concern. The birds within these groups are hypothesised to have strong relationships with their respective habitats and can be affected by disturbance and exotic plantations differently. For example, undisturbed forest type can attract a different bird assemblage that differs

from those in disturbed forests and in exotic plantations. This approach is important because it is likely to reveal bird species that are particularly sensitive, and habitat characteristics likely to be impacted strongly by various local human disturbances. Tropical forest managers need such scientific evidence for decision making and to help balance conservation and integrated management approaches (Bouvet et al. 2016).

1.3 Research aims and thesis structure

1.3.1 Research aims

The aim of this study is to determine the habitat characteristics of different forest types, some of these resulting from anthropogenic forest disturbance and exotic plantations, and the response of avian biodiversity to these characteristics in Afromontane forests. Specifically, it is to determine the difference in habitat characteristics in three forest types (natural disturbed and undisturbed forest types and eucalyptus plantation; hereafter forest types), and response of bird species diversity and community composition to forest types, and habitat characteristics in Mount Kenya's forests. This comes against the backdrop of an increasing human population, and access to and use of Afromontane forests from surrounding communities. The community access and use are either legally through permits and licenses as part of community participation and benefits (i.e., participatory forest management (PFM)) or illegally, leading to forest disturbance particularly at lower elevations of forests. There is also increase in exotic plantations within protected areas. As a result of these increasing pressures, the derived change in habitat characteristics and response of species is a significant and urgent knowledge gap in Afromontane forests.

To achieve the aims, the objectives are to determine:

1) Habitat characteristics across and within forest types, namely undisturbed, disturbed and plantation forests (Chapter 4).
2) Bird species richness and abundance of forest types (Chapter 5).
3) Bird community composition across forest types (Chapter 6)
4) The conservation and management implications of disturbance on avian biodiversity in Afromontane forests (Chapter 7).

1.3.2 Thesis structure

The thesis comprises of seven chapters, three of which are data chapters. All data chapters are set out in the style of a scientific paper. Each data chapter can be read either as a self-contained unit, or as part of the narrative whole. However, the introduction and methodology to each data chapter is supplemented by the general introduction, literature review and general methodological texts contained within Chapters 1, 2 and 3, respectively. Specifically, the thesis is structured as follows and as shown in Figure 1.3:

Chapter 1: General introduction, research context and rationale

This is the general introductory chapter with the study context and rationale. It is in this chapter that the general aims and objectives of the study is stated and sets out the thesis structure.

Chapter 2: Literature review

This chapter identifies existing knowledge gaps, and the position of the study among similar studies and why it is important.

Chapter 3: General methodology

This chapter describes the study area, study sites, sampling sites and sampling protocol. It introduces the general methodology of the study, data collection procedures and justification for all data variables collected. It also describes the general analysis and statistical approaches undertaken.

Data chapter, Chapter 4: Habitat characteristics within and across forest types

This chapter examines the variation/similarities in habitat characteristics within and across the three forest types in Mount Kenya forest. It addresses the questions:

1. *Is there more variation of habitat characteristics within or across forest types?*

2. *Is variation in habitat characteristics representative of forest classification into undisturbed, disturbed and plantation forests?*

Data chapter, Chapter 5: The value of undisturbed, disturbed and plantation forests for Mount Kenyan bird diversity

This chapter is used to describe the likely differences in bird species richness and abundance across forest types. Bird species or groups might associate with habitat characteristics, some strongly such that their presence (richness, abundance, or composition) may be predicted by or indicate the habitat conditions. Therefore the chapter further describes how species richness and abundance associates and is predicted by habitat characteristics.

The following questions are posed for this chapter:
1. *What is the overall bird species richness and abundance across forest types?*
2. *What are the species richness and abundance of (i) forest dependency (FD) birds (ii) birds in different dietary guilds (DG) and (iii) species of conservation concern (SCC) (threatened, endemic and restricted range species i.e., Afrotropical Highlands biome species (ATHB)).*
3. *How do habitat characteristics associate and predict bird species richness and abundance?*

Data chapter, Chapter 6: Bird community composition across forest types

The chapter describes bird community composition and how it varies across forest types.

The questions are:
1. *How does bird community composition differ across forest types?*
2. *What bird species are characteristic, shared or unique to a given forest type?*

Chapter 7: General discussion

The chapter discusses and synthesises the key results of the previous data chapters. It also discusses implications for conservation, study limitations and opportunities for future research.

To aid the discussion, the following questions are posed:

1. *What are the key results in relation to other findings in similar studies?*
2. *Based on habitat characteristics and disturbance, what are the conservation and management implications of the forest types in the Afromontane forests?*
3. *What are the conservation and management implications of the species richness and abundance and community composition of birds in different forest types in the Afromontane forests?*
4. *What are the opportunities for future research based on synthesis of key findings?*
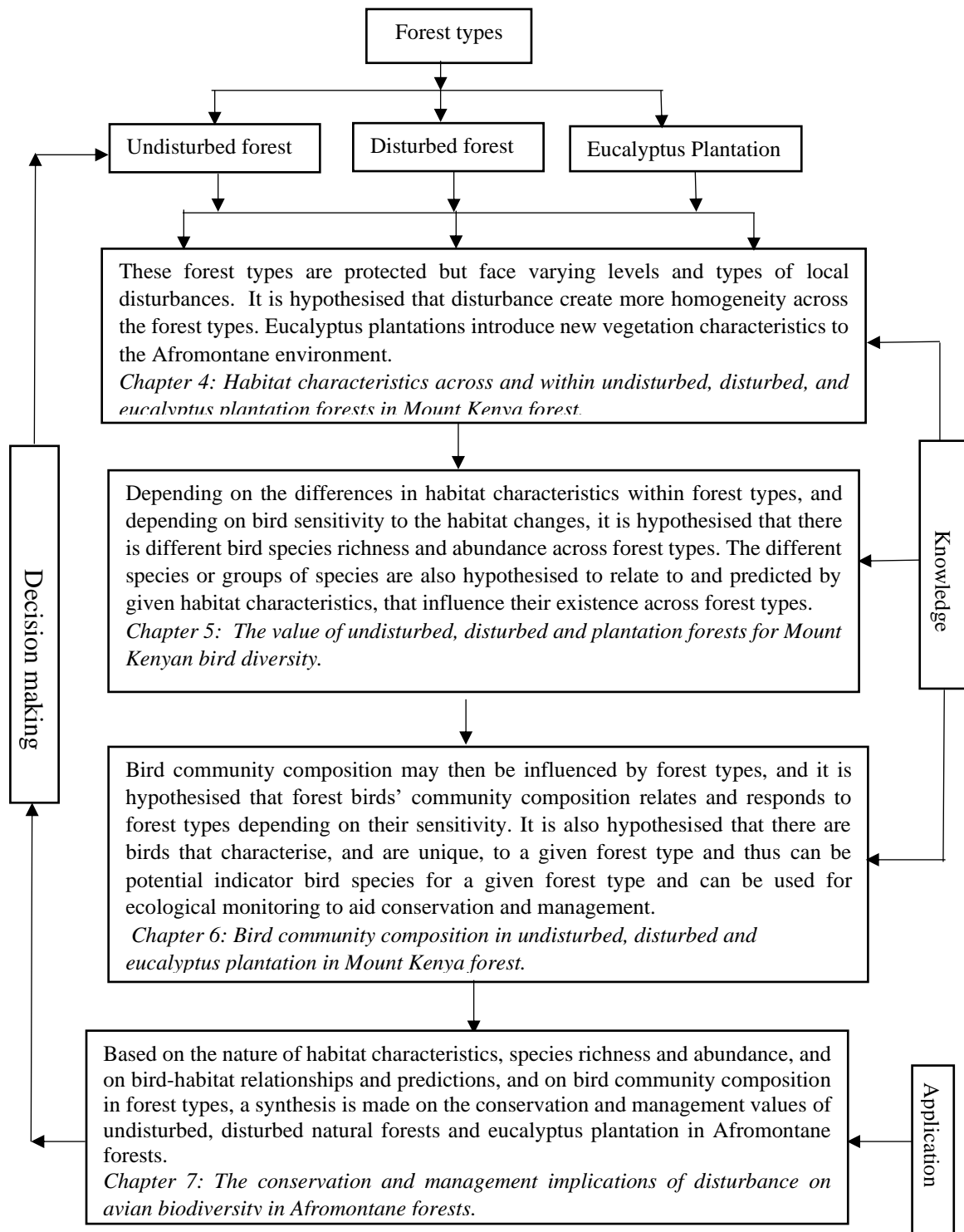
Figure 1.3: Schematic chapter arrangement and associated linkages between data Chapter 4 to Chapter 7.

Chapter 2- Patterns and drivers of tropical montane forest and birds

2.1 Importance of mountains for conservation

Mountains are recognised to be crucial for biodiversity conservation (Gerrard, 1990; Spehn et al., 2010). They constitute areas of high concentration of endemic species globally (Orme et al., 2005; Körner et al., 2005; Brooks et al., 2006), owing to biogeographic isolation under which mountain ecosystems have evolved (Barthlott et al., 2005). Mountains contain nearly half of the world's biodiversity hotspots, and support one-third of terrestrial species diversity (Myers et al., 2000; Körner, 2004), hence contribute immensely to terrestrial species conservation.

In the tropics, mountains are the richest areas for terrestrial biodiversity owing to habitat variability, marked orographic heterogeneity, varying climatic and microclimatic characteristics, speciation processes, and varying weather conditions (Dimitrov et al., 2012; Brown, 2014; Dulle et al., 2016; Rahbek et al., 2019). All these contribute to varied niches for biodiversity.

Only 9.8 % of the entire tropical forest biome lies within strictly protected areas (Schmitt et al. 2009). In addition, recent studies (i.e., Cronin et al., 2014; Tabor et al., 2018) revealed that tropical mountains are only minimally within protected areas and are not entirely protected against influence of human activities. Tropical mountains within protected areas, although protected, are affected by patterns of human activities particularly from communities adjacent to the forest areas (Wittemyer et al. 2008). The community members access the forest both legally (through participatory forest management (PFM) arrangements) (Matiku et al., 2012; Mbeche et al., 2021) and illegally to obtain forest resources such as honey, herbs, and fuelwood for their livelihoods (Wells et al., 2012). This presents a debate regarding how well mountains are protected by designations against negative effects of local disturbances, especially the mountain areas that are within lowland ecosystems that interact with the adjacent local communities (Payne et al., 2017).

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There is a potential risk of losing species through local disturbance in lowland mountain ecosystems areas, including those species that use the mountains as their last refuge, when the mountains are not protected effectively (Gratzer and Keeton, 2017; Elsen et al., 2020). This also includes species responding to effects of climate change (Elsen et al., 2018), that shift their range to higher elevations (Chen et al., 2011; Dulle et al., 2016; Batllori et al., 2017; Couet et al., 2022) where they are likely to encounter harsh environments not yet properly adapted to it (Huntley et al., 2006). The fact that the protected areas in mountain ecosystems, especially montane forests in the tropics are still small, and under constant encroachment and local disturbance (Crausbay and Martin, 2016; Edwards et al., 2019), pose greatest challenge to conservation of habitats and species dependent on those habitats. The existing conservation strategies in montane forests could be greatly enhanced if more of the risk factors such as human destructive actions, especially those encouraged by government policies and that potentially accelerate the loss of biodiversity in montane forests are well understood.

2.2 Contribution of tropical mountains for people and Participatory Forest Management (PFM)

Tropical mountains play critical roles in climatic regulation (temperatures and rainfalls), water cycles, ecological services, and protection against natural hazards such as flood controls (Price et al. 2011). This is through diverse vegetation in mountains that secures soils on steep slopes and contributes to the protection of landscapes and populations against natural hazards as well as the impact of extreme events (Körner et al., 2017). Mountain's ecosystems such as forests contribute to peoples' subsistence, welfare, and improvement of livelihood (e.g., Kono and Rambo, 2004) through direct utilisation of mountains resources such as medicines (herbs), and forest products of both wood and non-wood. This reliance on mountain ecosystems has been further supported and encouraged by government or institutional framework and policies (Lambin, 2014), such as PFM approaches (Wily, 2002). For example, in most east African forests, the forests are managed under PFM, which is a deliberate involvement of forest-adjacent communities in the management of forests within a structure that contributes to the local communities'

livelihoods (Himberg et al., 2009). With PFM approach, the communities living adjacent to forests are first required to be members of Community Forest Associations (CFAs). The CFAs then enters into collaborative management agreement with forest management authority after presenting a participatory forest management plan. Once permission is granted by the management authority to participate in the conservation and management of forest resources, CFAs are allowed to utilize certain forest resources through livelihood activities like bee keeping, grazing and sale of herbal medicine (Musyoki et al., 2022).

PFMs through CFAs, therefore, have allowed local communities and government agencies to join hands in forest conservation and management (Lambin, 2014), while benefitting local communities from forest resources. However, little thoughts have been put on possible detrimental extraction effects of CFAs members' practices on other mountain forest biodiversity (Jonsell, 2007; Bebbington et al., 2018). The effect of the access and resource use for example, on the integrity of mountain ecosystems, habitats and other biodiversity is still not clear especially at a local scale or microhabitat levels in tropical mountains. This is despite recommendation for more research across multiple scales to uncover novel ecological patterns or processes (Payne et al., 2017).

Tropical mountains are also of great religious, cultural and recreation significance (Fadiman, 1977; Sheridan, 2009). These make people pay homage to mountains as part of religious and cultural beliefs (Langdon, 2000; Price and Butt, 2000). People from urban areas seek solace in mountain environments as a respite from the stress of urban areas (Ali, 2002; Cetin and Sevik, 2016). Mountains also tend to represent the 'sacred' in many societies, giving them special status as holy sites (Langdon, 2000), and a focus for tourism activities (Funnel and Parish, 2005). All these activities can be legally acceptable or not in mountains, depending on protection status and existing policy arrangements of the mountain where it occurs, and depending on how it affects local habitat characteristics and other dependent organisms.

Human access and use of mountains and their ecosystems in the tropics are generally accompanied by actions that tend to negatively affect them (Körner, 2004). Rolston (1991) for example, pointed out that when humans recreate, they interrupt nature. He also

pointed out that humans hunt, fish, cut firewood, and harvest resources to a point where it is no longer sustainable. It has also been argued that when resources are taken for human benefits, wildlife, forests, species, and ecosystems suffer (Kaeslin and Williamson, 2010). Habitat characteristics and species are particularly threatened in mountains ecosystems under human access and use but the extent at which this is reflected within forest ecosystems in tropical montane forests remains largely unexplored.

2.3 Anthropogenic threats to tropical montane forests.

Tropical forest ecosystems host at least two-thirds of the earth's terrestrial biodiversity (Gardner et al., 2009) but their future is uncertain amidst arrays of threats from local human actions. These threats negatively affect the quality and the role of montane forests in fulfilling both the needs for humans and biodiversity to thrive (Asefa et al., 2017; Måren and Sharma, 2018). For example, the local communities have used the unique and globally significant Hyrcanian (Caspian) forests in northern Iran, for housing, farm development, and recreation (Zarandian et al., 2016), thus damaging its natural functioning. Both direct and indirect, internal, and external effects of human activities on protected montane forests are to blame for biodiversity loss (Cole and Landres, 1996; Asefa et al., 2017). Rapid population growth rate, poverty, limited land, and unemployment have been cited as the main drivers of deforestation in tropical countries (e.g., Iftekhar et al., 2003; KWS, 2010), that lead to the inability of the montane forests to provide critical ecosystem services.

Most tropical montane forests, even if protected, are surrounded by high anthropogenic related land-use intensities, and activities such as logging and establishment of monoculture plantations, that potentially can spill into protected forest and cause serious threats to biodiversity (Fahrig, 2003; Sodhi, et al., 2011; Newbold, et al., 2014; Asefa et al., 2017; Birdlife International, 2018; Måren and Sharma, 2018). The greatest threat to tropical montane forest is therefore through human activities (e.g., Cronin et al., 2014). Understanding the consequences of local human use of tropical montane forests is critical for creating strategies for contemporary forest management and conservation (e.g., Sanderson et al., 2002; Nepstad et al., 2002; Liu et al., 2003) in the face of local human

use. This involves determining how human caused disturbances in forest ecosystems functions and their ecological implications. This is useful to determine how impacts of human induced forest disturbances at a local scale affect habitat characteristic and how these influence the response of other organisms. The resultant knowledge, which is still lacking, is vital for how best to develop appropriate human use zones or buffer zones in tropical montane areas, and how to manage human activities as they continue to derive livelihoods from forests, and to ensure maintaining the integrity of the forest for biodiversity preservation.

2.4 Contribution of protected areas in tropical montane forest conservation

To address some of the threats to tropical montane forests, the Protected Area (PA) approach is still preferred as part of the modern strategy to conserve biodiversity (Dudley, 2008; Peach et al., 2019; Cazalis et al., 2020). This approach evolved from protectionists to current integration with community-based conservation approaches (MacKenzie, 1988; Adams, 2003). Studies have revealed that strategies that involved barring of people and prevention of exploitative use of resources and keeping other form of human activities to a minimum in PAs have not been effective (Brockington and Schmidt-Soltau, 2004). According to Anon (1994) and Gray et al. (2016), the PA approach is still very influential globally for conservation but has been integrated with human participatory management approaches like community-based management to safeguard the interest of local communities. This follows the observation that the future of much of tropical forest biodiversity (including on montane areas) depends more than ever on the effective management of human actors and their impacts on landscapes (Perfecto and Vandermeer, 2008).

Currently, PAs in montane forests play a role in ensuring that the livelihoods of local communities adjacent to the forests are enhanced through allowing them to derive some benefits from the forests (Naughton-Treves et al., 2005). At the same time, Pfeifer et al. (2012 (a)) and Måren and Sharma (2018) have illustrated how PAs have helped in maintaining and preserving biodiversity through prevention of spread of some illegal and destructive activities into mountains enclosed into PAs, maintaining their ecological

integrity. Şekercioḡlu et al. (2007) and Asefa et al. (2017) showed that PAs are important for conservation of particularly specialised species such as forest specialists, as compared to unprotected areas which are mostly heavily deforested. But these are not always the case as ecologists have also reported the negative human influence from the surrounding lands, that cross into PAs (e.g., Dasmann, 1988, Schonewald-Cox, 1988; Pfeifer et al., 2012 (a)). This influence can even extend from the PA periphery some distance into the protected areas (Revilla et al., 2001; Kolongo et al., 2006; Beche et al., 2022). The negative impacts may range from local-scale to landscape-scale impacts in PAs, with less severe impacts allowing for biotic resilience (Soh et al., 2019).

Other concerns arise particularly within African montane forests, where unlike elsewhere i.e., in southern America (Barrett et al., 2013) and southern Asia (Sodhi et al., 2004; Gibson et al., 2011), they are majorly threatened by small-scale logging and wood extraction, with disastrous long-term effects (Hosonuma et al., 2012). Yet there has been little research attention to address such local threats (Soh et al., 2019). These concerns in Afromontane forests are further compounded by the fact that the PAs' conservation and management policies and strategies currently in place, accommodate human use of PAs for extractive activities to fulfill socio-economic and cultural needs, without a clear understanding of the resultant ecological impacts. The protection and regulatory efforts by mandated conservation agencies to regulate human use of PAs i.e., through ranger patrols (Moore et al., 2018), Community-Conservation agency partnership agreements, and electric fencing (Massey et al., 2014), have been noted to be only effective in basic management activities (Bruner et al., 2001). This leaves substantial illegal human activities to infiltrate into PAs (Hansen and Defries, 2007), and thus human encroachment and disturbance remain issues of concern (Revilla et al., 2001). To further illustrate these anthropogenic disturbance and ineffective current management of it, Cronin et al., (2014) highlighted that the governments of Nigeria, Cameroon, and Equatorial Guinea designated PAs across tropical montane forest of Gulf of Guinea, West Africa, but are concerned about inadequate coverage of montane ecosystems, and ineffectiveness of management and regulatory enforcement to address anthropogenic pressure.

In conclusion, although PAs are still an essential element of any strategy to conserve tropical forest biodiversity, and probably the only means of safeguarding tropical forest species (Gardner et al., 2009), they are negatively affected by patterns of human activities within PAs and in adjacent areas (Wittemyer et al., 2008). Therefore, it makes ecological sense to assess the impact of local human activities on biodiversity within tropical montane protected areas, specifically on ecological elements of the PAs such as habitat characteristics and organism response to different forest types created by human disturbance. This also include investigation of potential ecological contribution of human environmental interventions like rehabilitation and restoration of PAs using introduced exotic plant species.

2.4.1 Community involvement in management and conservation of tropical montane forests

Tropical PAs face conservation challenges from the surrounding local communities who draw benefit from them (e.g., firewood, bushmeat, and clean water). PAs are regarded as areas of safety during times of human strife (Scherl et al., 2004) and local communities value them for their socio-economic welfare (Pattanayak, et al., 2003). Local communities benefit from PAs through consumptive and non-consumptive uses that can be obtained either illegally or legally through some agreed form of offtakes with the authorities (e.g., see Appendix 1). But when local communities are both responsible for extractive offtake of benefits as well as participating in conservation of PAs, they tend to do more extraction activities than conservation (i.e., De Sherbinin and Freudenberger, 1998; Bell, 2017). This presents a challenge on how well to balance conservation and local use of PA resources, particularly in areas that have attracted high population pressure around PAs (Luck, 2007).

It has been stated that the scale of human settlement around PAs is a strong predictor of illegal timber extraction (Karanth et al., 2006), fire frequency (Hudak et al., 2004), bushmeat hunting and general species extinction (Brashares et al., 2001) within PAs. The presence of these challenges might have necessitated suggestion of creation of multi-use buffer areas (or other zones) surrounding core habitats in PAs (Ebregt and Greve, 2000;

Martino, 2001). These buffer areas may facilitate effective protection and management of biodiversity while supporting potentially heavy human use from settlement next to PA borders (e.g., Martino, 2001; Wittemyer et al., 2008). This explains the presence of buffer zones, as a management tool, surrounding most of the PAs in tropical montane areas.

More than two decades ago, after apparent failures of other protective approaches to safeguard the integrity of PAs, there was a need to include buffer zonation and to involve local communities in decision making on conservation. It was observed that PAs in the tropics, and particularly montane forests, were highly likely to be well conserved if there was partnership between the communities and conservation non-governmental organisations (NGOs), protected area managers and public policy experts to address conservation challenges (De Sherbinin and Freudenberger, 1998). This was also part of the response to the need of instituting sustainable forest management, illustrated by the development of international policy initiatives such as the Convention on Biological Diversity (CBD), and the Forest Principles of Agenda 21. These had become a global environmental issue, reflecting widespread concern about high rates of forest loss and degradation (Newton et al., 2009). The benefits to communities and forests were centered on controlled access, extraction, and use of protected resources, accompanied by resource use and management agreements, negotiated within communities to regulate the resource use practices by individual members (e.g., Borrini-Feyerabend, 1996). Some of the suggested community involvement approaches included collaborative management of PAs, community-based conservation, and buffer zone development, all suggested to address high community resource dependencies from protected areas, and to ensure local communities support conservation efforts, while obtaining some negotiated benefits (e.g., Wittemyer et al., 2008). Creation of officially recognised buffer zones with restrictions on land use practices within it were part of the management approaches (Ebregt and Greve, 2000), where the buffer zones are where community members are allowed to derive benefits, and not to put pressure on the core zones (Rotich, 2012). These approaches have since been implemented in tropical montane forests (e.g., Tole 2010) some more than 2 decades ago in Africa.

The attractiveness of the collaborative and community-based conservation approach arose from the fact that scientists widely accept it as suitable (Berkes, 2007; Mureithi et al., 2019). Conservation biologists and resource managers show interest in management arrangements involving local communities and the use forest resources for community benefits (e.g., Panayotou and Ashton, 1992; Hladik et al., 1993). This is especially where local communities were subject to be involved in decision making either as partners with managing authorities e. g. government /NGOs (Smoke, 2003), or as independent entities managing biological resources for their benefits (Grootaert and van Bastelaert 2002). It is also believed that with judicious use of PA resources, rural income can be enhanced without degrading forests, traditional knowledge can be used in conservation, and local communities are willing to participate in conservation because of the economic stake they are to gain (e.g., Larson and Ribot 2004). However, access and over extraction of resources for example in buffer zones, resulting from these arrangements, can have a negative impact on quality of habitats and status of biodiversity (Kideghesho et al., 2013; Muhumuza and Balkwill, 2013). Researchers also question the conservation potential of extractive resources (see Larson, 2002; Kideghesho et al., 2013; Crawhall, 2015; Görmüş, 2016; Walde, 2019). For example, contrary to the belief that the above approaches help in conservation of resources and benefit the local communities, Bergl et al. (2007) and Oates (1999) argue that community-based conservation projects increase pressure on protected areas in West Africa and bring distraction from overall conservation goals. In fact, some are reported to have done more harm than centralized forest management (Larson, 2002; Ribot, 2003; Sunderlin et al., 2005). However, the impact of extraction of forest resources on forest tree structures and general habitat characteristics is unknown in most areas where community involvement resource use is currently being practiced.

2.5 Forest ecology

2.5.1 Forest structure and its conservation values

The ability of the forest to influence the local climate and microclimate, and to provide resources for other biodiversity depends on its structure (Messier et al., 2013). It is known that a building unit of a forest is a tree with its characteristics of heights, branches, and

cover, giving tree its three-dimensional structure (Shugart et al., 2010). A tree is composed of living and non-living components that contribute to its structure and grows extending vertically and horizontally, forming a structure suitable to offer unique niches for organisms. Kimmins (2009) describes forest structure as the collection of interacting species of trees and other organisms that forms an ecosystem consisting of an array of living and non-living components (soil, water, air, temperature) as influenced by topography, climates, and microclimates. He described that the variations in vertical and horizontal structure of forests across the landscape (e.g., as illustrated in Figure 2.1), allows for the interaction of biotic and abiotic factors to synergistically shape species distributions, composition, and suitability of habitats to offer conditions and resources required by other species.
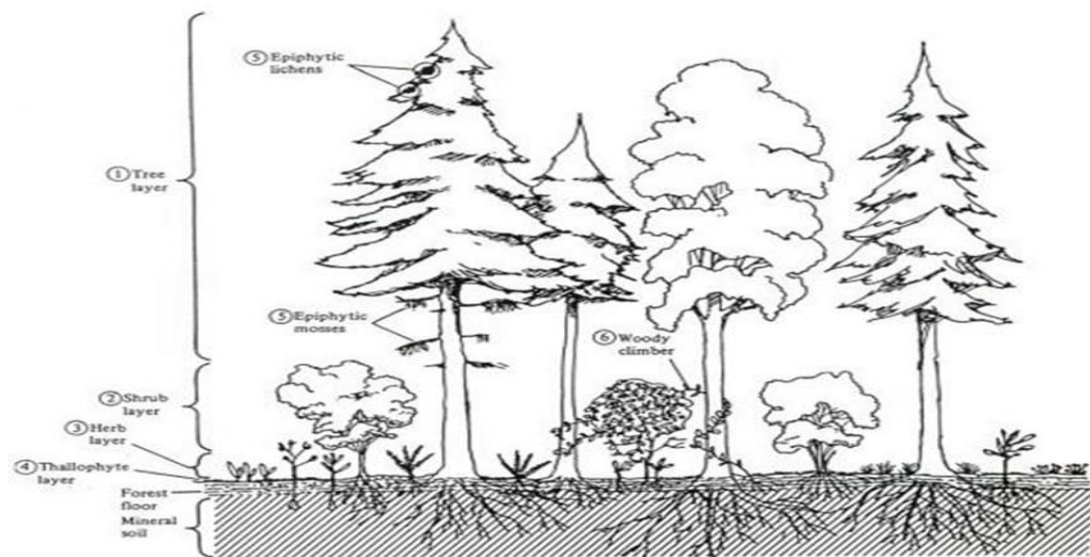
Figure 2.1: Forest community showing vertical and horizontal structure, individual growth characteristics, and height growth potential (Source: Kimmins, 2009).

The components of a tree: the branches, bark, leaves, roots, living and dead parts (snags), makes shelter, roosting and cover for forest organisms. The dead stumps, cavities, and root holes, all create microhabitats for yet more organisms in a forest, where they can live, forage, shelter, cache food and stalk their prey (Vander Wall, 1990) (i.e., Figure 2.2 a, b.). In addition, snags provide high quality habitat and food resources for dead-wood dependent invertebrates (e.g., Figure 2.2 c). According to Attua and Pabi (2013), trees

must be constantly monitored and managed to direct successional processes towards maintaining species and habitat diversity.
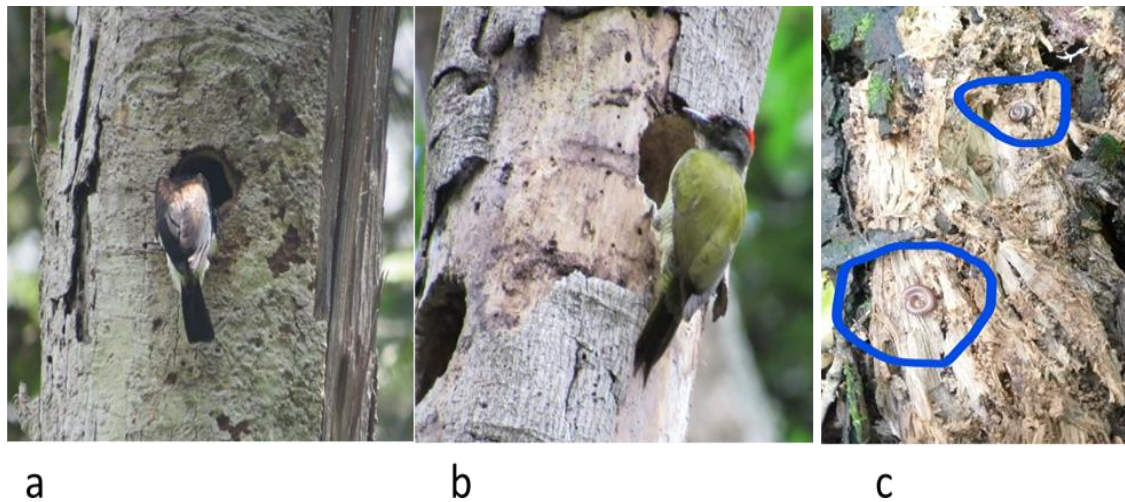
Figure 2.2: (a) white-eared barbet (*Stactolaema leucotis*), (b) Fine-banded woodpecker (*Campethera tullbergi*) foraging and using snag cavity nest, and (c) Centipedes (highlighted in blue circles) on rotting wood trunk. Photos taken by author during the study on 17/10/2019, 14/4/2019 and 12/3/2020 respectively in Mount Kenya forest.

Ecologically important tree features are removed when selective logging, fuelwood collection, and livestock feeds are harvested, or other local uses are extracted from the forests. Deadwood as illustrated in figure 2.2 has been identified as a significant contributor to habitat provision in forest ecosystems and is recognized as an indicator of forest health (Brockerhoff et al., 2017). Deadwood tends to be more abundant in old growth, primary undisturbed forest, or less managed stands and provides both habitat and forage for a large suite of forest biota (Seibold et al., 2015) (i.e., Figure 2.2 b, c). In addition, many forest birds and mammal species depend on the presence of tree cavities associated with deadwood for nesting and roosting (Cockle et al., 2011; Burgar et al., 2015) (Figure 2.2 a, b). However, the ecological contribution of deadwood is lost when it is removed from forests as part of fuelwood or other uses.

2.5.2 Importance of tropical montane forest ecosystems to birds

Tropical montane forests form one of the ecosystems in which birds have formed very close and intricate relationships with their habitats. They have evolved tight symbiotic

relationships i.e., flower-pollinator symbiosis, seed dispersal and with their prey (Whelan, 2001, Peh et al., 2005, Stratford and Şekercioḡlu, 2015). These could be the reasons why some are 'tied' to the forests, with some of these birds having been classified as forest specialists, while others are generalists (e.g., Bennun et al., 1996). These groups (specialists and generalists) are negatively affected differently with any form of forest disturbance, and probably respond differently to it. Some forest birds can probably be driven out of their habitat when disturbance results in habitat degradation. Tropical montane lowland forests which receive greater incidences of human disturbance also support a large proportion of species threatened with extinction such as forest specialists (Stratford and Şekercioḡlu, 2015; Brockerhoff et al., 2017; Birdlife International, 2018). Given that there is also high human dependency on these forests, it makes the tropical lowland forest areas suitable for studying the effect of human activities on forest birds.

The most important contribution of tropical montane forest ecosystems to birds is perhaps the provision of the essential resources necessary for the completion of their life cycles. This could depend on habitat quality such as how it provides food for adults and breeding sites i.e., for nestlings and nesting sites (see Martin and Joron, 2003; Stratford and Şekercioḡlu, 2015). Birds breeding success may be associated with the quality of habitats provided by forest structural characteristics such as structural arrangement from forest floor to the canopies (e.g., Bakermans, et al., 2012). For example, canopy openness has been found to influence reproductive success of forest birds (Norris et al., 2004). That is, canopy cover regulates the temperature for chicks during their development (Dawson, et al., 2005). Habitat structural arrangement of forest trees can be affected by any form of local disturbance related to removal of habitat characteristics e.g., tree lopping (Seidler, 2017). Furthermore, forests can provide microclimates and shade that help birds tolerate physiologically challenging temperatures (Schooler et al., 2020; De Frenne et al., 2021). Small passerines, for example, move into trees and forested habitats during dry seasons in the tropics to get cool refugia when temperatures are high enough to cause thermal stress (Seavy, 2006). Removal or degradation of forest conditions thus may determine the presence and persistence of birds in such ecosystems in relation to amount and conditions of habitat characteristics they require.

2.5.3 Drivers of bird distribution, diversity, and composition in tropical montane forests

Seasonal factors

Seasonality plays a major role in determining the abundance and distribution of birds in the tropics among other taxa (Kluge et al., 2006; Williams and Middleton, 2008; Mengesha et al., 2011; Girma et al., 2017). This is because seasonality affects food and cover availability for birds, which in turn affects breeding and ultimately survival (Mengesha and Bekele, 2008; Mengesha et al., 2011). For example, feeding guilds of birds demonstrate seasonal variation along an elevational gradient in the Himalaya Mountains (Katuwal et al., 2016) and in Ethiopia (Girma et al., 2017) while forest regeneration has been linked to rainfall (Pfeifer et al., 2012 (b)). Vegetation structure also changes in response to seasonal rainfall and temperature changes (Abera et al., 2018). For instance, vegetation becomes more open during the dry season as compared to during the rainy season in the tropics and thus may influence bird distribution in relation to predation risks in foraging areas (Whelan and Maina, 2005; Renner et al., 2012).

Seasonality therefore can influence the temporal dynamics of bird species richness and composition (Shiu and Lee, 2003) and abundance of migratory species (Girma et al., 2017). Despite seasonality evidently playing a role in affecting habitat conditions and characteristics in the tropics, the extent to which it influences the habitat dynamics in montane forests is yet to be fully appreciated, especially when it is combined with the effects of disturbances.

Topographic factors

Elevation and slope have been found to affect vegetation structure, site productivity, distribution, composition, and secondary biotic interactions (Waterhouse et al., 2002). Species richness and the composition of bird communities often change rapidly with elevation (e.g., Blake and Loisselle., 2000). Some studies have found elevation to influence the assemblage of species of conservation concern, and some that are observed to associate with specific habitat characteristics (e.g., Zou et al., 2012). In Himalayan

Mountains for example, elevation explained the occurrence of river associated birds along the rivers (Manel et al., 2000). Elevation has also been found to be a major environmental factor contributing to variation in the species richness trends of residential bird assemblage in Taiwan (Shiu and Lee, 2003). Elevation is responsible for structuring bird communities in the tropical Andes and the most important variable explaining bird species composition there (Montaño-Centellas and Garitano-Zevala, 2015). Slope is a good predictor of bird richness in Wondo forest, Ethiopia (Girma et al., 2017). Slope is also a predictor of mean species richness per point count in Comoro Island montane forests (Monticelli, 2012). Nevertheless, how slope influences and predicts Afromontane forest bird diversity and community composition is yet to be fully appreciated.

Habitat characteristics

Habitat characteristic arrangements (such as vegetation structural arrangements, human disturbance related characteristics) of sites affects species richness, diversity, or abundance (Verschuyl et al., 2008; Gumede et al., 2022). For example, complex vegetation stratification, foliage density and canopy complexity owing to associated diverse niches, may provide essential foraging, roosting, and nesting requirements for rearing of bird offspring (Augenfeld et al., 2008), including cover against predation. Bird studies have revealed that vegetation related structural variables such as foliage height, connectivity, heterogeneity, density of understory vegetation, and cover can influence abundance, diversity and populations through provision of conditions and resources (Gabbe et al., 2002; Waterhouse et al., 2002; Tews et al., 2004; Whelan and Maina, 2005; Goetz et al., 2007; Crampton, 2011; Blendinger et al., 2012; Ndang'ang'a, et al., 2013; Casas et al., 2016). Bird abundance, diversity and species composition is therefore likely to vary to a certain degree in different areas in response to degrees of change in vegetation structure (Aleixo, 1999; Gumede et al., 2022). The condition of forest vegetation structure can either support occupancy of some species, have neutral effects on some, or have negative effects on others (Banks et al., 2017).

Disturbance regimes (i.e., difference in disturbance across a contiguous forest) could result in occurrence of different vegetation structure or habitat characteristics especially

between highly disturbed and moderately disturbed or undisturbed forest sites. This could include difference in characteristics such as canopy cover, tree size, and tree height differences. These vegetation structures are responsible in mediating climatic effects on bird species richness (Ferger et al., 2014). On a microclimatic and micro-habitat scale, bird species may respond to vegetation structure variation i.e., from habitat edges to interiors (Whelan and Maina, 2005), and may either be attracted or repelled depending on their required suitable conditions. This response pattern may disappear or diminish when understory vegetation characteristics are removed or reduced in forests (through disturbance), destabilising the former suitable conditions.

Habitat vegetation characteristics, for example class size distribution of trees (normally expressed as measure of diameter at breast height (DBH)) affects bird abundance, diversity, and richness (Waterhouse et al., 2002; Brown, 2008). This class size distribution (distribution of different stem sizes of trees) is targeted by selective loggers in tropical montane forests (Wimberly and Spies, 2001), notably through pole cuttings. Therefore, habitat characteristics that form important components of forest ecosystems are facing local disturbances and yet rarely get researchers attention, particularly determining how such disturbance affects organisms in tropical montane forests.

It has been revealed that birds' response to change in vegetation characteristics contributes to the understanding of habitat selection and biodiversity conservation (Yuan et al., 2014). For example, certain species, such as a single species or a selected few, have been chosen to represent an entire assemblage (e.g., Githiru, et al., 2007), and have resulted in findings capable of application, where a conservation approach or management protocols can then be developed based on the needs of the surrogate species (Banks et al., 2017). In tropical montane forests that are under increasing pressures from human activities, and where the exact and specific habitat characteristics affected is unknown, a study is needed to reveal the forest structural characteristics impacted and how these affect other organisms. It is critical to understand these ecological variables and how they contribute to spatial distribution of avian species diversity (Huang, et al., 2014). Previous studies (e.g., Banks et al., 2017; Girma et al., 2017) have recommended in-depth studies of the effects of

vegetation characteristics and responses of groups of species to establish optimal protocols for wildlife management in forest reserves.

Anthropogenic disturbance and habitat degradation

Anthropogenic habitat destruction in natural forests is amongst the main drivers of biodiversity loss in the tropics (Morris, 2010; Alroy, 2017). There is increasing demand for forest related products, timber, and charcoal from forests in tropical montane forests (Taucher et al., 2020). From these demands, there is likely habitat characteristic changes that indirectly affect the abundance, composition, and distribution of vertebrates (e.g., Leyte-Manrique et al., 2019). Human activities thus shape biodiversity patterns and forest ecosystem processes in these areas (Magurran and McGill, 2011).

Extreme disturbances are likely to destroy a community, but some disturbances (moderate) may enhance biodiversity by releasing resources or by promoting the coexistence of species adapted to different conditions (Connell, 1978; Valladares et al., 2015). Therefore, it would be beneficial to ascertain the extent of disturbance and how it enhances or degrades biodiversity. If possible, anthropogenic disturbance that maintains or actions that lead to an increase in biodiversity, even if it includes some habitat perturbation, can be encouraged as part of the ecosystem management. But anthropogenic disturbances that lead to habitat degradation and loss, that affects trophic organization and ecosystem functioning (Gray et al., 2007; Newbold et al., 2015), should be controlled. Most habitat disturbance involving alteration of vegetation structure and fragmentation such as livestock grazing, forest fires, selective logging, and hunting (i.e., as illustrated in Figure 2.3), has been regarded as a major threat to tropical forests (Şekercioḡlu, 2002 a; Heikkinen et al., 2004; Gray et al., 2007). In most cases, the intensity of these destructive activities progress from forest edges towards forest interiors (direction of human actions in Figure 2.3 below). This leaves a continuum of forest disturbance intensities, with most disturbed forest parts being those close to forest edges, while relatively undisturbed or intact forests lie towards forest interiors.
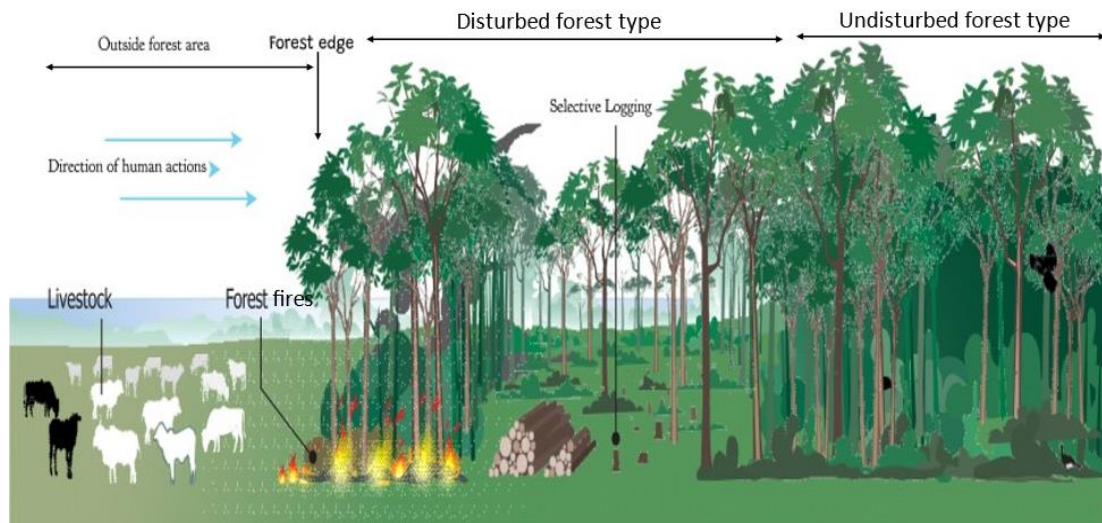
Figure 2.3: illustration of anthropogenic disturbance and how it affects tropical forest habitat structure from forest edge to forest interior. Barlow et al. (2016) report that the combined effects of various human-driven disturbances in the forests of the Brazilian Amazon can cause biodiversity losses on a scale like, or greater than, those caused by deforestation alone. Conversion of forest to farmland, forest fires, selective logging can affect forest structure with areas near edges being most vulnerable. This can result in direct and indirect biodiversity loss. (Source: Adapted and modified from Bartlow et al., 2016).

Extractive human disturbance interferes with the tree structural arrangements through removals and affects ecosystem, biomass, and habitat suitability for other species (Malhi et al., 2018). For example, if a tree with a lot of leaves is lopped, its vegetation (leaves) become less, and microhabitats within it are exposed. The cover for nesting birds is removed and the usefulness of such a tree for other organisms reduces significantly. In addition, if a standing snag is harvested for fuelwood, the cavity nesters and birds that depend on foraging on snags (e.g., see Figure 2.2 in section 2.5.1) will be reduced at those sites. Therefore, small-scale microhabitat disturbance affects bird's habitats. In particular, small-scale (local scale) habitat characteristics can readily be overlooked because they tend to be cryptic (Peres, et al., 2006), yet are wide-spread and rarely studied in tropical forests (Gutzwiller and Anderson, 1999; Kirika et al., 2008).

Human disturbance to tropical forests differs depending on the type and intensity (Montaňo-Centellas and Garitano-Zavala, 2015) and the type/nature of forest involved

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(Agramont et al., 2012). It also affects different bird species differently (Şekercioḡlu et al., 2002 b; Harris and Pimm, 2004; Sodhi et al., 2004), with different species responding to it differently (Verhulst et al., 2004; Lee et al., 2005; Gray et al., 2007; Kirika et al., 2008). Sensitive bird species (such as forest specialists) have been found to be influenced by changes in canopy cover (Lee et al., 2005) and loss of food (Gray et al., 2007), and thus modification of these characteristics through local anthropogenic disturbance can change species richness and community composition among forest birds (Kirika et al., 2008; Montaňo-Centellas and Garitano-Zavala, 2015). Anthropogenic disturbance affects individual birds by exposing them to predation risks or by reducing their potential refuge sites to escape from predators (Whelan and Maina, 2005). It also affects the ability of forest regeneration from the loss of forest bird seed dispersers (Kirika et al., 2008), thus affecting the health of the forests.

Villages (with inherent differences in socio-economic, and cultural characteristics) are expanding in tropical areas, and local populations are increasing near PAs in tropical montane areas (e.g., Wittemyer et al., 2008), such that it is speculated that anthropogenic disturbances negatively affect biodiversity within the PAs. This is following high demands for, easy access to, and minimal prohibition on the use of protected forests by local communities, who can do selective logging, harvesting of medicinal plants, pasturing cattle and forest fires leading to habitat modifications, especially on the forest floor (Popradit et al., 2015; Barlow et al., 2016). For example, local uses make forest canopies more open, thus allowing more light penetration into the forest floor, creating suitable conditions for shade-intolerant and less competitive vegetation, including invasive species to grow (Wagner et al., 2011). There will then be high growth of near-ground thickets or high shrub density of vegetation (e.g., Ngueguim et al., 2018), changing the original characteristics and conditions of forests, and affecting organisms such as birds. These changes may happen closer to forest edges or can extend to varying distances into the protected area interiors (Hansen and Defries, 2007; Laurance, 2000). How negatively or perhaps positively these disturbances affect the ecology of different forest sites or habitat characteristics and birds in tropical montane forests have not been investigated. To the best of available knowledge, there is no ecological research that has investigated the habitat structural characteristic differences between disturbed and

undisturbed contiguous natural forest sites in the tropical montane forests. More research is therefore needed to understand the role of anthropogenic disturbance in tropical montane forests.

Introduction of exotic plant species in the form of plantations is widespread in tropical forests (Denslow and DeWalt, 2008), and positively correlates with human disturbance (Fine, 2002), particularly in Afromontane forests (Teucher et al., 2020). Exotic plantations are used to regenerate degraded forests and for afforestation (Farwig et al., 2008). Expansion of these plantations is expected to continue due to continued growing demands for tree products such as timber by growing populations (Calviño-Cancela, 2013), while on the other hand, natural forest will continue to be destroyed and lost, only replaced by exotic plantations.

Despite their exotic nature, plantations have been found to play ecological roles to some extent (Brockerhoff et al., 2008; Volpato et al., 2010; Wu et al., 2016). Compared to natural forests or mixed-species forests, planted forests usually have a lower level of biodiversity (e.g., Şekercioğlu, 2002 a; Barlow et al., 2007; Brockerhoff et al., 2008; Farwig, et al., 2008), and their suitability to provide ecosystem services is reduced relative to natural forests (Dhanya et al., 2014; D'amato et al., 2017). In addition, some forest bird specialists tend to be more sensitive and respond negatively to habitat conversion from native to exotic plantation forests (Farwig et al., 2008). All these effects could be because of habitat structural differences between exotic plantation forests and natural forests. However, comparative studies that focus on specific habitat characteristic differences between natural forests and exotic plantations have not been carried out. More information is therefore needed to fully understand the habitat structural differences of exotic plantations and differently disturbed natural forests and their roles in conservation (Coote et al., 2013).

2.6 Use of birds as a taxon of focus in tropical montane forest

Birds are one of the most important ecological assets in tropical montane forest due to the following. First, they play an important role in key ecological services such as acting as predators and prey. They work as seed dispersers and pollinators in the maintenance of

ecological processes of forests. Secondly, they are very sensitive to changes within their habitats (Sodhi et al., 2005; Kumar and Shahabuddin, 2006; Yap et al., 2007) hence, can easily respond to slight environmental changes. Thirdly, they are mobile species, increasing the likelihood that they can reach wider environments, habitats, and can experience and respond to variation in resource availability. Birds, therefore, form the most suitable study taxon owing to:

(i)   being easiest and inexpensive taxa to study (easy to sample).

(ii)   providing a quick and spatially efficient way to assess habitat conditions.

(iii) information on birds being commonly available.

(iv)   taxonomically well-known and easily censused.

With these characteristics, birds have high potential to be used as an effective biodiversity indicator because: (a) of ability to respond much faster to disturbance (i.e. sensitive to habitat characteristic change) than other indicator species (Ramírez-Soto et al., 2018), and (b) they have been found to be the most cost-effective for monitoring as Ecological-Disturbance Indicator Species (EDIS) relative to other organisms such as small mammals, bats and leaf-litter lizards (e.g. Peck et al., 2014). Researchers such as Lambert and Collar (2002), and Sigel et al. (2006) have considered birds as good indicators of habitat type and conditions in forests.

2.7 East African montane forests

There are several East African montane forests covering a total area of 65,500 Km^2. These mountains range from northernmost Mount Kinyeti in the Imatong Mountains of Southern Sudan, through Mount Moroto in eastern Uganda, Mount Elgon on the Kenya-Uganda border, to the one in the east and west of the eastern Rift valley (Figure 2.4). These include Aberdare range, Mount Kenya, Mount Kulal, Mount Nyiru, Bukkol and Nguruman Escarpment in Kenya. In Tanzania there are Mount Kilimanjaro, Mount Meru, Ngorongoro and Marang forest in northern Tanzania.

Among these montane forests, there are three which are the highest and only glaciated mountains i.e., Mount Kilimanjaro (5, 895 m) in Tanzania, Mount Kenya (5, 199 m) in Kenya and Ruwenzori (5, 110 m) in Uganda.
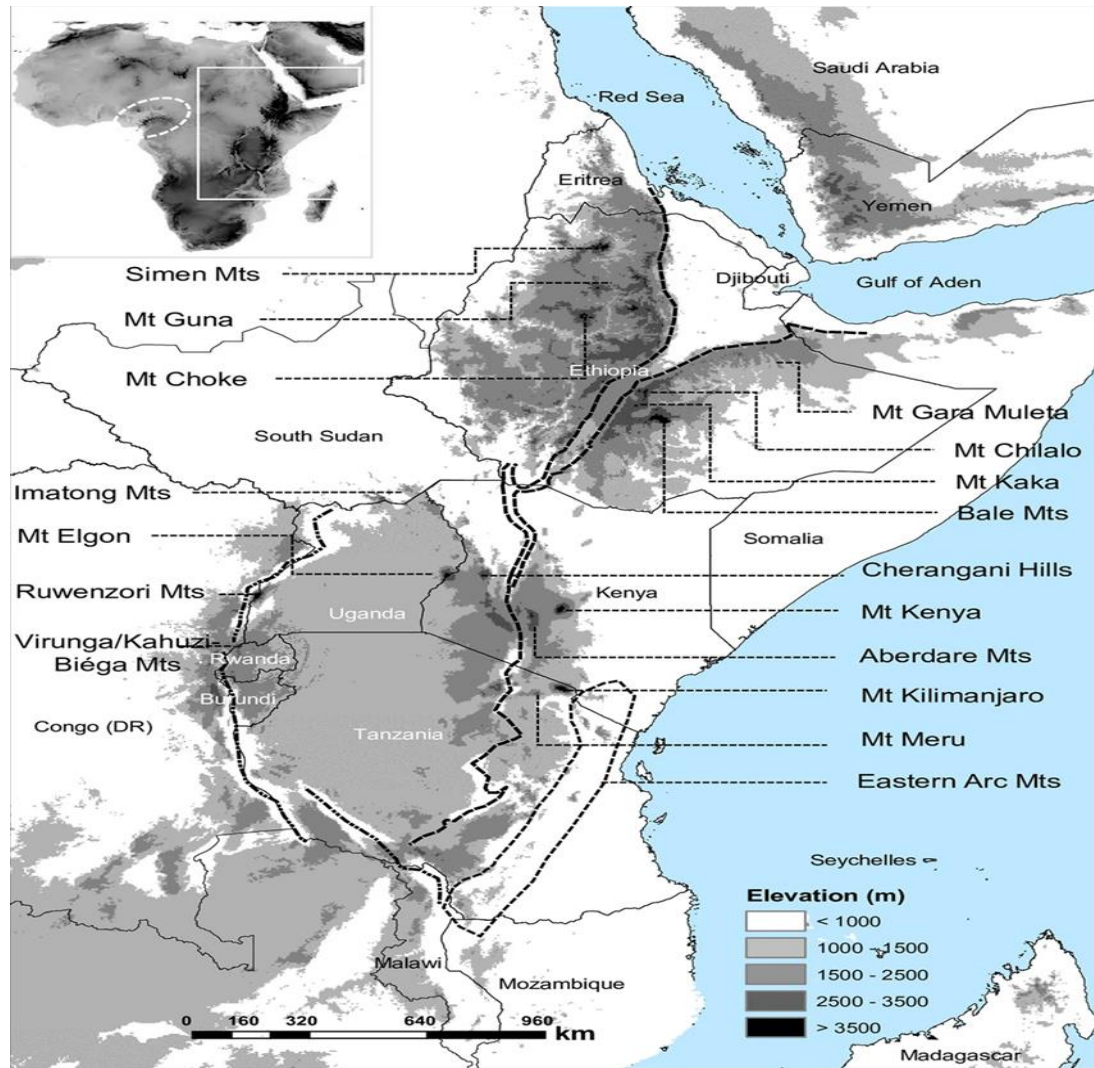
Figure 2.4: East African mountain distributions containing montane forests in South Sudan, Ethiopia, Kenya, Uganda, and Tanzania. Adapted from Brochmann et al. (2021).

These mountains and their forests all serve important local, regional, and national economies by providing ecological services and socio-economic roles (UNEP, 2012; Jones, 2014). For example, they are all considered "water towers" and support recreational and tourist attractions. In addition, the forests within them have a high concentration of plants and animals, some of which are endemic i.e., Eastern Arc Mountain hotspot sites (BirdLife International, 2012).

The natural montane forests in east Africa are increasingly receiving global attention following their role in maintaining a global share of biological diversity (Taylor, 2015).

They provide migrant birds with wintering grounds, and maintain threatened, restricted-range, and endemic species threatened by climate change (Dulle et al., 2016). However, these forests are among the most extensively threatened by human activities (Balmford et al., 2001; Newmark, 2002). High human population, agricultural expansion, fuelwood and timber extraction for construction, overgrazing and fires are the main significant drivers of forest change in East African tropical montane forests (Kigenyi et al., 2002; Lung and Schaab, 2010). Yet, there is poor research representation on impacts of habitat degradation on tropical montane biodiversity (e.g., Soh et al., 2019).

2.7.1 Characteristics of Kenyan montane forests.

Kenya covers an area of 582 646 km^2 of which 2008 km^2 (3.4 %) is covered by forests. These forests are of different variety: lowland rain forest, dry coastal forest, and montane forests (Peltorinne, 2004). They are broadly grouped into natural forests (1700 km^2), exotic plantation forests (122 km^2), forests in private farms (woodlots) (124 km^2) and the rest are mangrove forests (Peltorinne, 2004; UNEP, 2009). Most of these forests are fragmented and degraded to various degrees. According to UNEP (2009), an estimated 2.9 million people live within five kilometers from indigenous closed-canopy forests and derive 70 % of their income from Kenyan forests (also Wass, 1995).

There are five montane forests in Kenya. These are Mount Kenya, the Aberdares, the Mau Forest complex, Mount Elgon, and the Cherangani Hills, in total covering 2 percent of the country (Ahmed and Mlay, 1998; Peltorinne, 2004). They are the main sources of major rivers and referred to as Kenya's 'Water Towers', supplying 75 % of surface water (Kenya Forest Service (KFS), 2010; Crafford et al., 2012). These forests provide other products and services for industries and households and are surrounded by dense populations of farming communities due to the surrounding area's suitability for agricultural practices. The local communities depend on these forests for traditional cultural needs, recreation, and tourism amenities (Adams, 2012).

 For biodiversity conservation, these forests hold species of conservation concern, some endemic, and others of restricted-range, especially birds (Bennun and Njoroge, 1999). All these montane forests are Important Bird Areas (IBAs) (Bennun and Njoroge, 1999).

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However, they are being degraded through increasing population pressures and encroachment, resource extraction, logging, settlements, overgrazing, and charcoal burning (Gathaara and Leakey, 1999; KWS, 2002; Eshiamwata, 2012; Teucher et al., 2020). Commercial over-utilisation of trees in the lower forest zones present a great threat to biodiversity.

The government has been issuing restrictions and regulations to control forest destruction following overutilisation (Teucher et al., 2020). For example, several national laws, regulations and bans in public forests were put in place in the past as well as recently (e.g., Makanji and Mochida, 2004; Muisyo, 2018). These include a presidential logging ban on indigenous timber that has been in effect since 1998, the Kenya Forests Act 2005 that was reviewed and passed in 2005, forest (charcoal) regulations that were put in place in 2009, and the Forest Conservation and Management Act (Kenya Gazette Supplement, 2016) that became operational in the year 2016. More recently, the Kenyan Ministry of Environment and Forestry declared a nation-wide logging ban extension declared in 2018 (Muisyo, 2018) for all public and community forests that is still in force as of the year 2022. Even with all these in place, forests in Kenya continue to be degraded through human activities.

Mount Kenya and its forest is designated as an IBA, among other biodiversity and conservation importance. But it is still relatively unexplored in terms of scientific research in comparison with other mountains in East Africa (e.g., Newmark, 1991; Newmark, 1993; Hitimana et al., 2004; Soini, 2006; Bett et al., 2016; Dulle et al., 2016; Schellenberger et al., 2017). Not much is known on the impact of human disturbance on birds, habitat characteristics and their relationship with birds. The only studies that have tried to scientifically investigate aspects of Mount Kenya birds are Tattersfield et al. (2001), Ndegwa (2014), Kioko et al. (2016) and Mahiga et al. (2019). Two of these are unpublished thesis and a technical report. It might be argued that given the spatial scales and remoteness of Mount Kenya, and the vastness of ecosystems involved, it is not surprising that quantitative studies in Mount Kenya is still minimal and yet robust information from here for hypothesis testing is still needed. The challenge that remains is to understand the contribution of anthropogenic disturbances on integrity and

conservation values of Mount Kenya forest, and the value of exotic plantation and how likely it is to offer additional habitats for birds.

The main existing threats affecting Mount Kenya forest include small scale farming that extend to the boundaries of protected areas, forest product removal such as placement of hives and associated destruction (Figure 2.5 a), both legal and illegal removal of grazing material (Figure 2.5 b and e), illegal logging (Figure 2.5 c and d), fuel-wood collection (Figure 2.5 f), cultivation (*shamba system*), and water abstraction that continues to be obtained by people from Mount Kenya forest.

Figure 2.5: (a) A beehive placed on a tree. Preparation for hive placement involve cuttings and removal of vegetation (b) and (e) Community member(s) transporting livestock feeds harvested from protected forest (c) and (d) illegally logged tree in disturbed forest site in Mount Kenya (f) firewood collected within disturbed forest in Mount Kenya. Photos taken by researcher during the fieldwork on 11/10/2018 (a), 13/09/2019 (b), 8/11/2019 (c), 7/02/2019 (d); 17/10/2019 (e), 17/01/2019 (f)).

These human activities may influence the ecological integrity of the habitats and affects the status of birds in Mount Kenya forest. Therefore, this study attempts to understand the human disturbance and habitat characteristics and response of bird communities to forest degradation and eucalyptus plantations in Mount Kenya.

CHAPTER 3 - General methodology

This chapter provides details of the study area and general methods used during this research. Methods and statistical analyses that relate to specific results presented in this thesis are outlined in subsequent data chapters.

3.1 Study area: Mount Kenya

Mount Kenya is located in central Kenya and straddles the equator, approximately 190 km northeast of Nairobi and 480 km from the coast (0' 10'S; 37' 20'E; Figure 3.1).
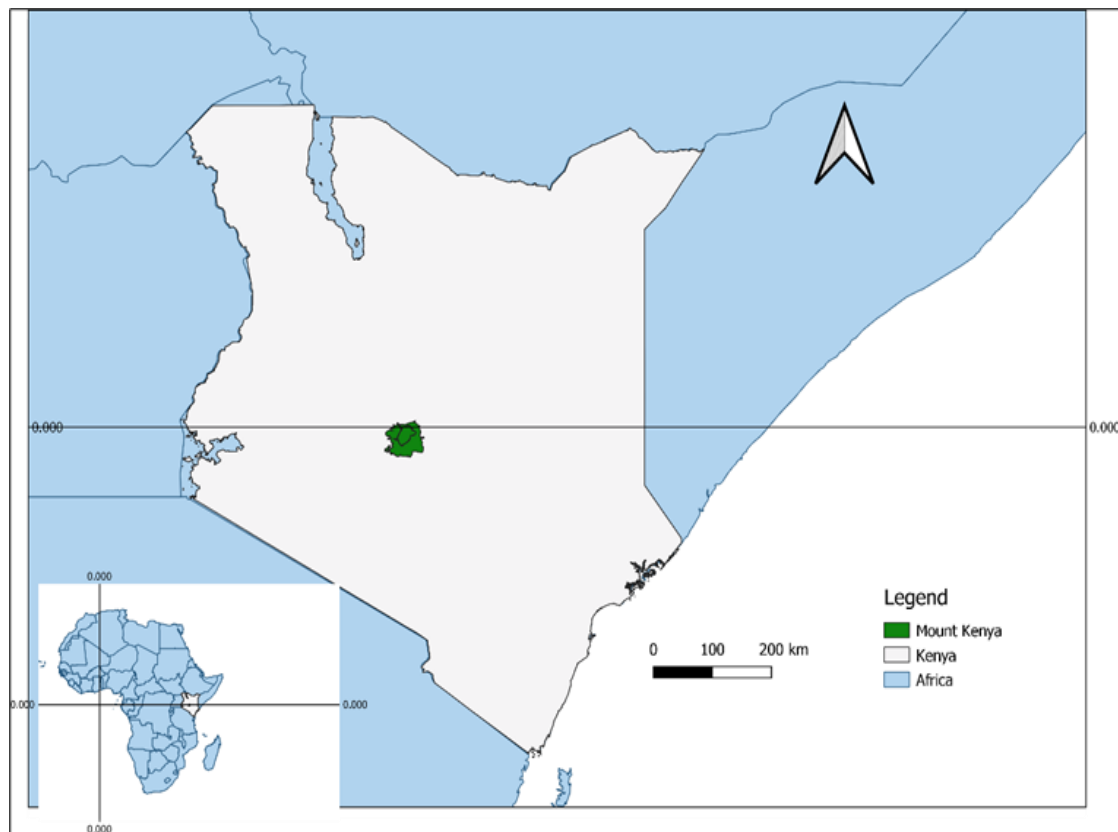
Figure 3.1: Geographical location of Mount Kenya within African continent (inset).

Mount Kenya is an extinct volcano formed between 2.6 and 3.1 million years ago (Speck, 1982). Elevation ranges from a low of 1,500 m above sea level (asl) to a high of 5,199 m asl, making it Africa's second highest mountain, after Mount Kilimanjaro in Tanzania (Gathaara and Leakey, 1999). The Mount Kenya Forest Reserve (70,520 ha) was gazetted

in 1932 and this was not revoked when the same area was gazetted as a national reserve in 2000 under the Wildlife Conservation and Management Act. This forest is therefore managed jointly by KFS and KWS, respectively. The Mount Kenya National Park (71,510 ha) was demarcated within the Forest Reserve's upper zone above 3200 m asl in 1949 and is managed by KWS. In 1978 the national park was designated a UNESCO Biosphere Reserve. The national park and the forest reserve, combined, became a UNESCO World Heritage Site in 1997 (142,000 ha)

3.1.1 Climate

Mount Kenya has a typical equatorial mountain climate. Annual precipitation varies from a low of about 870 mm at the base of the mountain to about 1,970 mm at the peak and temperature from about 12°C to -4°C (Zhou et al., 2018). Large daily temperature fluctuations occur that are greatest on the lower slopes and can be as high as 20°C (Nyongesa and Vacik, 2019).

The year is divided into two distinct wet seasons and two distinct dry seasons (Camberlin et al., 2014). The *long rains* occur between mid-March and June, followed by the wetter of the two dry seasons which lasts until September. The *short rains* last between October and December, and, finally, the driest season occurs from January to mid-March. The south and south-east slopes receive an average rainfall of 2300 mm annually, while an average of 900 mm is received in the northern and north-west parts (Gathaara and Leakey, 1999).

3.1.2 Vegetation zonation

The steep climate gradient along Mount Kenya's slopes results in a dramatic change in vegetation cover with altitude and distinct zonation (Zhou et al., 2018). The lower slopes are composed of montane forest that can be divided into lower montane wet forest to the northeast, east, southeast, and southern part of Mount Kenya, and the lower montane dry forest to the west (Figure 3.2) (Zhou et al., 2018). Lower montane forest is a natural forest dotted with exotic plantations in some parts of it, particularly in degraded sites and in sites without trees. The lower montane wet forest starts at a lower altitude of about

1450 m asl in the eastern and southeastern parts of the mountain than in the lower montane dry forest in the western parts which start at 1850 m asl (Zhou et al., 2018). This is followed by bamboo forest before transitioning to upper montane forest, ericaceous, paramo and nival zones (Niemela and Pellikka, 2004; Zhou et al., 2018) as illustrated in Figure 3.2.
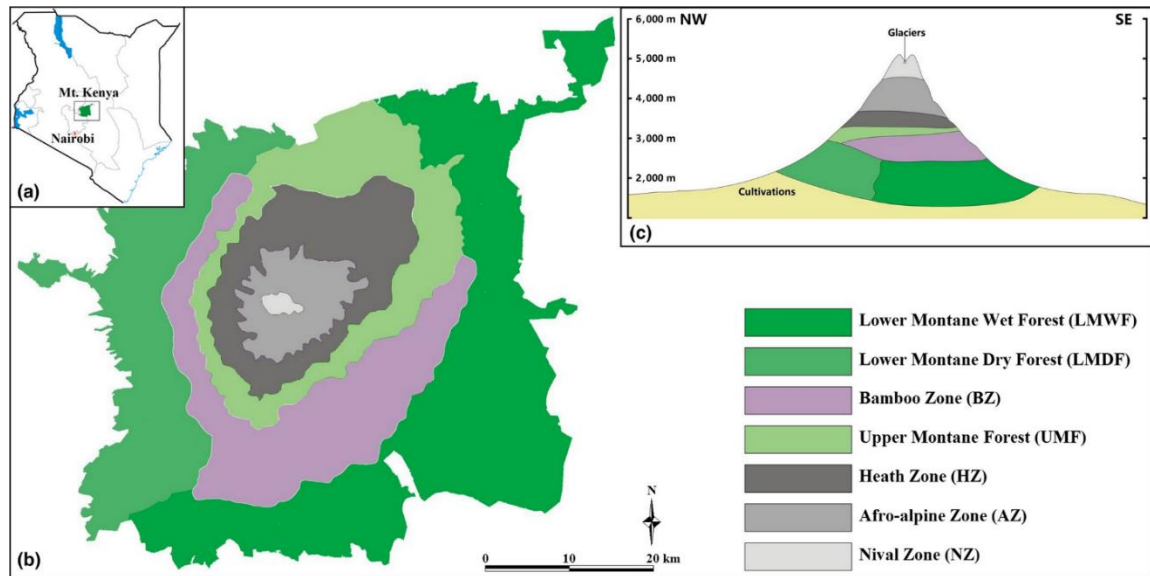
Figure 3.2: Vegetation zones of Mount Kenya. (a) The location of Mount Kenya in Kenya (adapted from Zhou et al., 2018). (b) The vegetation zones of Mount Kenya in top view (adapted from Niemelä and Pellikka, 2004; Zhou et al., 2018) (c) The vegetation zones of Mount Kenya from northwest to southeast in lateral view (adapted from Coe, 1967, Niemella and Pellikka, 2004, and Zhou et al., 2018).

Natural forest

Natural forest covers an area of 139,424 ha (approximately 57 % of the mountain) (KFS, 2010). This comprises the lower montane (wet and dry) at an altitude between 1450-3110 m asl, bamboo zone at between 2140-3270 m asl, and upper montane forest zones at between 2650-3890 (Zhou et al., 2018) (Figure 3.2). The lower montane wet forest from altitude of 1200-1500 m asl is covered with evergreen mountain forests characterised by *Newtonia buchananii, Lovoa swynnertonii* and *Chrysophyllum gorungosanumare* as prominent tree species (Bussmann, 2006). On the south and south-eastern slopes at altitudes between 1500-2400 m asl are moist Ocotea forest (*Ocotea usambarensis*) while the lower and middle sub-montane region it is dominated by evergreen species of

Syzygium guineense (Myrtaceae) and *Aningeria adolfi-friederici* (Sapotaceae). In the east of Meru in the lower Imenti forest occurs Newtonia forest on the eastern slopes at lower altitudes. *Croton sylvaticus-Premna* forest occurs in the upper Imenti forest near Meru at altitudes of 1500-1800 asl m and *Croton-Brachylaena-Calodendrum* forest also occurs near Meru at altitudes 1450-1850 m asl. In the lower montane dry forest, the lowermost part of the western side there is evergreen submontane semi-deciduous forest, where *Calodendron capense* or *Croton megalocarpus* are common. There are also woodlands dominated by *Acacia drepanolobium* from 2000-2300 m asl.

The upper montane forest is dominated by montane rainforest zone, at altitude of 2000-2400 m asl on western side and 2400-2700 m asl on the north-eastern slopes; generally dominated by *Podocarpus latifolia* mixed with *Nuxia congesta* at the upper altitudes. On the northern part, the woodlands lead directly into xeromorphic forests, which are almost entirely dominated by Pencil Cedar (*Juniperus procera*) and Wild Olive (*Olea europaea subsp. africana*) (Bussmann, 2006).

The bamboo zone occurs both as pure stands and mixed with indigenous trees (KFS, 2010). Pure bamboo (*Arudinaria alpina)* stands occur between 2550 and 2650 m asl on the southern and eastern side. Mixed bamboo and indigenous tree stands are dominated by *Arudinaria alpina* and extend from 2500 to 3200 m. On the western side, the zone lies between 2560 and 3200 m asl while on the eastern side it lies between 2800 and 2950 m asl.

Plantation forest

Exotic plantation forests within the Mount Kenya Forest Reserve date back to the early 1900s, when commercial forest plantations were initiated between 2200 m and 2400 m asl (KFS, 2010). The main purpose was to supply commercial forest products to the forest industries such as Nithi timber cooperative society limited, Irangi timber industries, and Mount Kenya sawmills among many others (Lieth and Lohmann, 2013) located within the forest adjacent areas (KFS, 2010), as demand for timber and other wood products

could not be fulfilled by the natural forests alone. The main commercial tree species planted as plantations included exotic trees such as Cypress (*Cupressus spp.*), Pines (*Pinus spp*), and Eucalypts (*Eucalyptus spp.*). In addition, there are also native tree plantations, mainly of *Vitex keniensis* and *Juniperus procera*.

Currently, the exotic plantations mostly occur on the peripheries of natural forests and are patchily distributed in the eastern and south-eastern of Mount Kenya, and mainly concentrated in the perepheries in western part (Figure 3.3), all totalling approximately 18,618 ha (KFS, 2010).
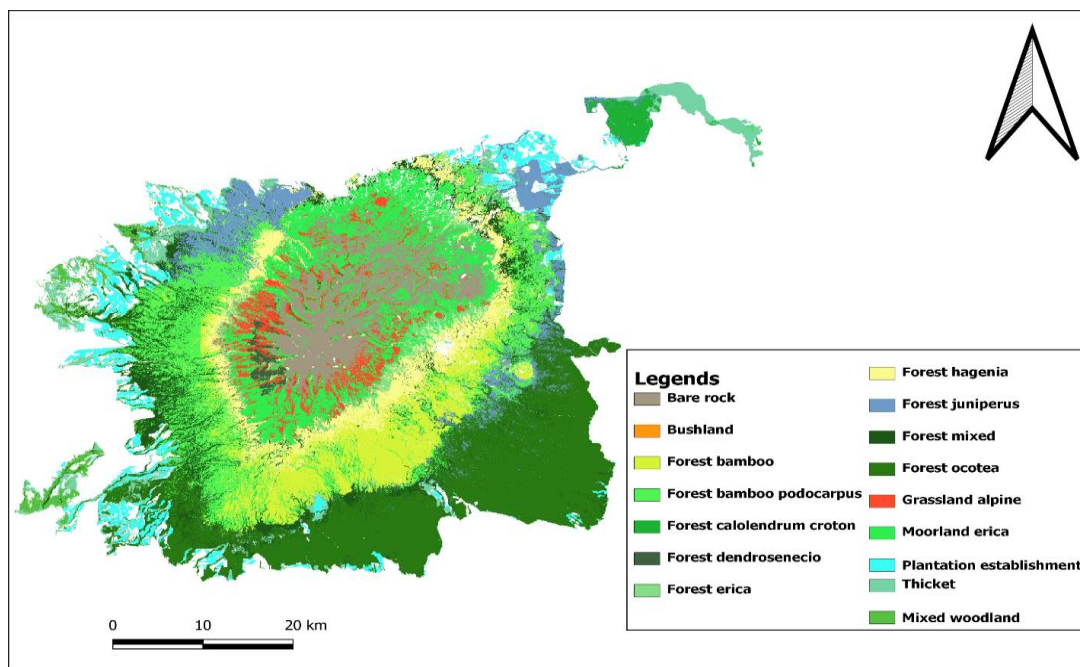
Figure 3.3: Distribution of vegetation types within Mount Kenya, with exotic plantation on the lower peripheries of the Mount Kenya forest reserve. Most exotic plantations are on the western part of Mount Kenya with patchy distribution in southern, southeastern, and eastern Mount Kenya dominated by eucalyptus species.

Due to growing demands and government policies, some of the plantations were introduced in the 1960s between 1550 m and 2400 m asl to supply commercial forest products, and others were planted in the 1980s and 1990s to restore destroyed and degraded areas due to heavy logging and forest fires, fuelwood, and for windbreaks (KFS, 2010; Lieth and Lohmann, 2013). Eucalyptus plantation for example dominate high

agriculturally potential areas for their use in tea factories as fuelwood. Plantation forests are also being expanded both within protected forest reserves and surrounding community lands in order to meet the requirement of the Kenya Forest Act 2005, Constitution of Kenya 2010, and Kenya Forest Policy 2014, that require an increase in forest cover to at least 10 % of the country's total land area (MEWNR, 2014). Cypress tree plantations mainly dominate the north, north-western, western and south-west of Mount Kenya's forest, while eucalyptus tree plantations dominate the eastern, south-eastern, and southern parts (Figure 3.3).

3.1.3 Land use in areas adjacent to Mount Kenya forest reserve

The areas outside and adjacent to Mount Kenya forest reserve are mainly characterised by smallholder tea farms, subsistence crop production and livestock keeping (Emerton, 1999; Willkomm et al., 2016). In the eastern, southeastern, and southern part of Mount Kenya where rainfall is relatively high, and suitable for agricultural activities, the agricultural farms, including tea farms extend up to the forest edge demarcated by electric fence (Figure 3.4).
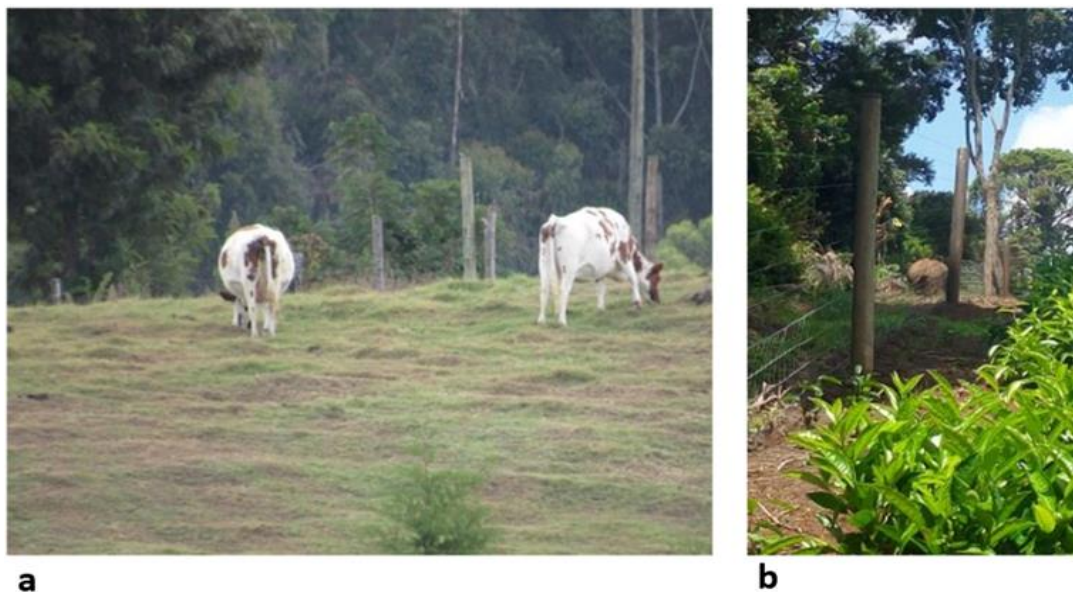
Figure 3.4: Land use in areas adjacent to Mount Kenya forest reserve. (a) Livestock grazing outside but next to forest edge in Ruthumbi. Photo taken on 04/05/2019 (b) Tea farm established to the edge of the forest demarcated by electric fence next to Chuka forest station. Photo taken on 28/11/2018 during study area reconnaissance. Phot credit: Author.

One main driving factor for agriculture in this area is its climatic and soil suitability for agriculture production. These areas are highly populated by people (Emerton, 1999), with the current population density that ranges from 153- 413 persons/km^2 in counties surrounding Mount Kenya forest reserve (KNBS, 2019), and which constraints the individual farms to be very small. The farms are mainly for agricultural production, focusing on the production of tea, coffee, maize, beans, potatoes, vegetables, dairy, and mixed livestock production (Emerton, 1999). The population adjacent to Mount Kenya forest reserve normally supplement the small farm production with non-timber forest products such as foraging of wild fruits, hunting of game meat, fishing, collection of honey, firewood collection, charcoal making and herbal medicine collection.

3.1.4 Management of Mount Kenya forest reserve

Mount Kenya forest reserve occurs between 1450 m asl to 3200 m asl, above which lies Mount Kenya National Park (Zhou et al., 2018). It is managed by KFS through the implementation of its Mount Kenya forest reserve management plan with the aim of forest conservation and development (KFS, 2016). This includes establishment of plantations in the place of harvested indigenous (natural) stands, regulating access to resources under participatory forest management (PFM) with local communities (i.e., harvesting of livestock feed, grazing in protected forest, beehive placement, fuelwood collection etc (see appendix 1), and sustaining a forest industry (KFS, 2010; KWS, 2010). KFS also guide the establishment, development, and sustainable management, including conservation and rational utilisation of the forests and allied resources for socio-economic development (KFS, 2010). Since it is also a national reserve, KWS also jointly with KFS manage some aspect of the forest reserve, particularly through the implementation of the now concluded Mount Kenya Ecosystem Management Plan (2010-2020) (KWS, 2010). KWS have the key purpose of protecting and conserving biodiversity, especially endemic, rare, and threatened species (KWS, 2010). To achieve their key mandates, KWS and KFS, in their respective management plans, have established largely complementary/overlapping zonation schemes to aid in the implementation of activities towards their goals. The KFS zonation scheme is termed Forest Reserve Management and

Utilization zonation, while KWS zonation scheme is termed Mount Kenya Ecosystem (MKE) zonation scheme and is detailed below.

Forest Reserve Management and Utilisation zonation (Kenya Forest Service)

The Mount Kenya Forest Reserve Management Plan (KFS, 2010) developed zonation based on vegetation types. It has three major zones within the Forest Reserve (namely, natural forest, plantation forest and Nyayo tea belt), and one zone outside of the protected forest called the 'community intervention zone' to address the activities being undertaken in the farmlands (Table 3.1). The demarcated zoning is therefore based on activities and developments allowable in different parts of the ecosystem based on management objectives implementation (KFS, 2010).

Table 3.1: Mount Kenya Ecosystem zonation as recognized by Kenya Forest Service as part of their Mount Kenya Forest Reserve Management Plan 2010-2019 (KFS, 2010).

Zone	Sub zone	Criteria	Management objective
Natural forest	Protected areas	-Nature reserves -Biodiversity hot spots -forest wetlands -Moorland	• Protect ecological integrity of the protected areas • Preservation of the water catchment function • Ecological research and education
	Conservation	Natural forest, bamboo and glades not zoned as protected areas	• Restoration of degraded forest areas • Preservation of the water catchment function • Development of ecotourism & Nature based enterprise • Controlled utilization of wood and NWFP[*] • PFM† activities. • Ecological research and education
Plantation	NA	All areas designated for commercial forest production.	• Commercial production and extraction of wood and NWFP[*] • PFM[†] • Commercial forest management research
Nyayo Tea Belt	NA	100 metres belt in tea growing areas	• Commercial tea growing • Fuel wood plantations for internal consumption
Community intervention	NA	Farmland within 5km from the Forest Reserve boundary.	• Promote on-farm tree growing • Promote income generating activities • Support community institutions in forestry programmes • Protection of riparian belt and hilltop afforestation

[*] Non wood forest products † Participatory forest management

Mount Kenya Ecosystem zonation scheme (Kenya Wildlife Service)

The MKE zonation scheme as per KWS management plan resulted partly from the work of Gathaara and Leakey (1999) and recommendations from a report carried out to investigate the changes in the state of conservation of Mount Kenya's forest (Vanleeuwe et al., 2002). It covers the entire MKE and comprises six visitor use zones (KWS, 2010), as shown in Figure 3.5 and as detailed below.
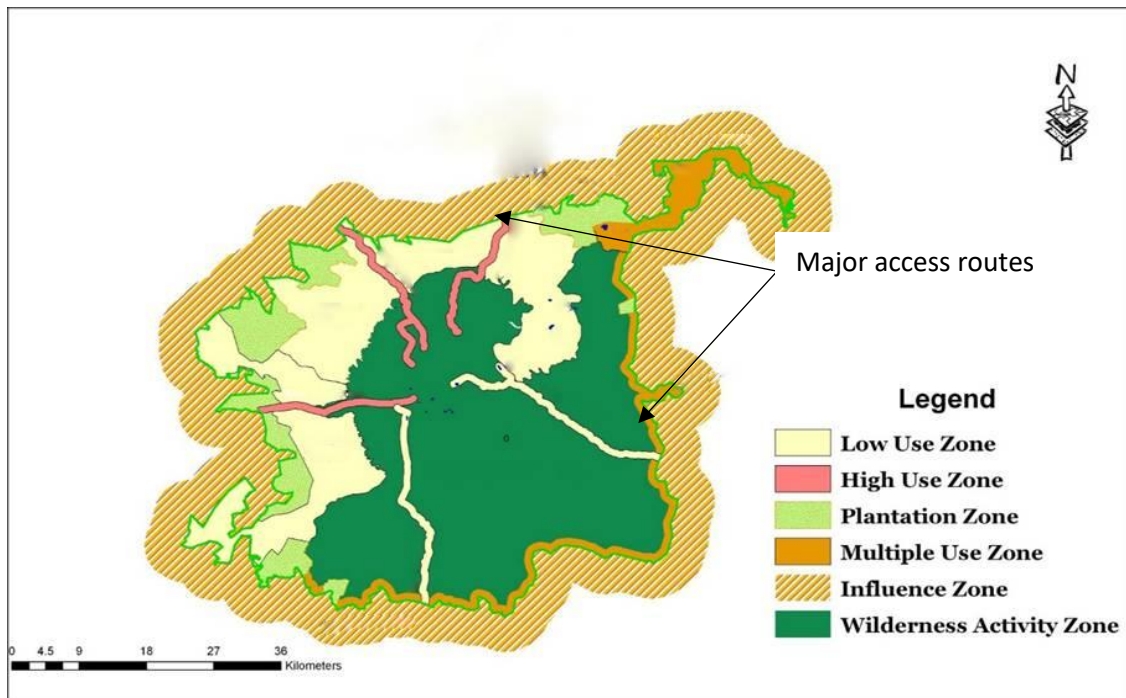
Figure 3.5: Different visitor use zones within the Mount Kenya Ecosystem, developed as part of the Mount Kenya Ecosystem Management Plan 2010-2020 (adapted from KWS, 2010).

i) Wilderness activity zone

This KWS zone overlaps with the KFS's natural forest zone and is a continuous large extensive area with minimal human interference (representing undisturbed forest area). This zone covers a major part of Mount Kenya National Park, high altitude parts and the natural forest types.

ii) Low use zone

This zone comprises a 500 m buffer either side of unpopular hiking routes that attract a low number of visitors in eastern and southern Mount Kenya (i.e., Chogoria and Kamweti (shaded yellow routes) respectively in figure 3.5), and wildlife viewing areas at the northern moorland and west (Bongo Sanctuary sub-zones) (shaded yellow in Figure 3.5). The low use zone was meant to offer a low level of visitor use using these routes.

iii) High use zone

This zone comprises a 500 m wide corridor on either side of the three most popular and heavily used mountain hiking routes – two routes in the western region (Naro Moru and Sirmon) and one route in Marania on the leeward side of the mountain.

iv) Multiple-use zone

This zone covers an area of about 330 km^2 and comprises a continuous belt of about 1 km in width into the forest reserve from the reserve boundary/forest edge on the eastern and south-eastern side of the mountain (Figure 3.5). The management aim of this zone is to provide wildlife migration corridors and an effective buffer for the natural forest against human related threats by engaging forest-adjacent communities in participatory forest management activities. It is part of the natural forest zone and conservation sub-zone shown in Table 3.1, and KFS termed it as conservation area use zone in its management plan (KFS, 2010), and provides controlled permitted multiple uses, including tourism, apiculture, harvesting forest products (i.e., non-wood forest products, livestock grazing, collection of medicinal plants, firewood collection, and harvesting of building materials. Consequently, this zone can be defined as disturbed forest (in comparison with forest in the wilderness zone that is relatively undisturbed).

v) Influence zone

This zone lies outside the forest reserve and overlaps with the community intervention zone under KFS's Mount Kenya Forest Reserve Management Plan (KFS, 2010, Table 3.1). It comprises community land within a 5 km belt from the reserve boundary, which

is demarcated by an electric fence (but with several entry points for permitted community members). The influence zone is community owned land where members practice varied farming and other socio-economic development activities. Community members in this zone are mainly targeted by KFS to be actively involved in Mount Kenya forest reserve management through Community Forest Associations (CFAs) under the umbrella of participatory forest management. The community members are mainly encouraged to engage in conservation friendly ventures like tree planting, soil conservation and farm based-bee keeping easing human-use pressure on the adjacent protected forest reserve. However, with controlled permission, KFS allows members of the community (who are members of the CFAs) within the influence zone/community intervention zone to access selected benefits from the forest reserve such as apiculture, collection of firewood, harvesting of selected forest products and livestock grazing materials. These also may represent possible main conservation challenges to the protected forest reserve.

vi) Plantation Zone

This zone is demarcated for sustainable production of timber through the KFS's Plantation Establishment for Livelihood Improvement System (PELIS) programme, and production of firewood, fibers, carving wood or any other wood for which there is demand. The main trees under plantation include exotic cypress, pines, and eucalyptus, although there have been plantation stands of native *Grevellia* spp. The northern, western, and south-western parts of Mount Kenya (which are also the low rainfall receiving regions) are where most exotic plantations are located, dominated by Cypress spp, covering 16,575 ha – 89 % of the total exotic plantations in Mount Kenya. The eastern, southeastern, and southern forest plantations are patchily distributed and total 2,043 ha (i.e., 11% of Mount Kenya's plantations), and are mainly dominated by Eucalyptus spp (KFS, 2010).

3.2 Study sites and forest types

3.2.1 Study sites

This study is confined to three zones of the MKE, namely the wilderness activity zone, the multiple-use zone, and the plantation zone (KWS, 2010). The selection criteria for the

study sites were defined during the reconnaissance period between October and November 2018, and before the pilot study that started in December 2018. Specifically, study sites were those that encompassed the following:

i) Were within the lower montane vegetation zone of Mount Kenya forest reserve, in the eastern, southeastern, and southern part of Mount Kenya forest reserve, coinciding with lower montane wet forest and areas where members of the local community interact with the forest and are dependent upon its resources. This is because the study tries to investigate how local community disturbance affect forest habitat characteristics and its biodiversity.

ii) Borders the community intervention zone/influence zone, with similar or comparable socio-economic activities since these may influence the way local communities use the surrounding natural forest resources (Al-Subaiee, 2016; Garekae et al., 2017).

iii) Contain mature growth of eucalyptus plantations (i.e., more than 10 years since planting) of sufficient area coverage to accommodate placement of sampling stations.

iv) Similar climatic conditions to each other (i.e., temperature and precipitation).

v) Comparable or small altitudinal range.

Out of the eight forest stations (forest stations are forest management units, each under forest manager in a given forest sector with management staff) that formed possible study sites in the eastern, south-eastern, and southern parts of Mount Kenya (i.e., Castle, Chehe, Chogoria, Chuka, Irangi, Kangaita, Meru, and Ruthumbi), three met all of the above listed criteria, namely: Castle, Chuka and Ruthumbi forests.

The three study sites (Figure 3.6, Table 3.2) are surrounded by high concentrations of farming communities with comparatively high population density, estimated at 400 persons/km^2 and steadily growing annually as compared to the northern, western, and southwestern part of Mount Kenya (Emerton, 1999). The sites receive a high annual rainfall of an average 2300 mm. Tea, coffee and livestock keeping dominate at the three sites as the main economic activities among the surrounding communities. These

communities wholly depend on Mount Kenya forest reserve for their water supply, wood, and non-wood forest products, e.g., firewood/fuel wood, timber, honey harvesting/beekeeping, medicinal herbs, and fodder for their livestock (Emerton, 1999).
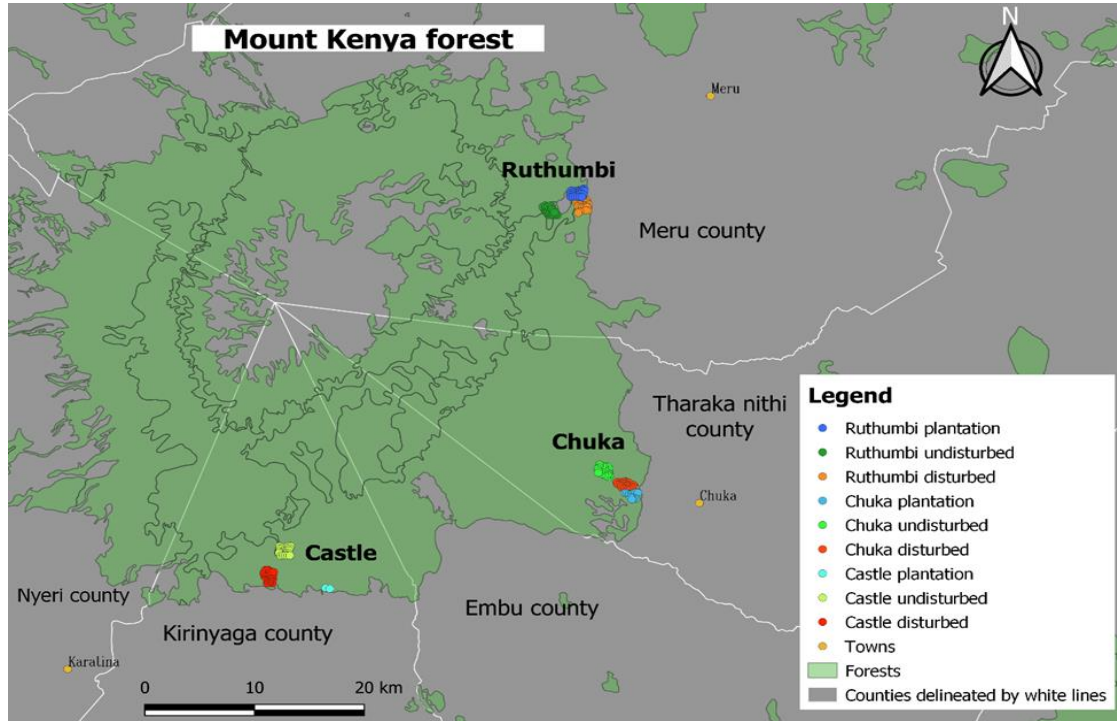
Figure 3.6: Location of the three study sites - Castle, Chuka and Ruthumbu - in the Mount Kenya Forest reserve. The dots in each study sites represented with different colours per study site and forest types represent sampling point count stations.

Table 3.2: The size of the natural (both undisturbed and disturbed) and eucalyptus plantation forests in the three study sites (KFS, 2010).

Study site	Total forest area (Ha)	Natural forest (Ha)	Plantation forest (Ha)
Chuka	23,403	17,603	200
Castle	19,971	7,526	576
Ruthumbi	12,605	6,435	365
Total	**55,979**	**31,564**	**1,141**

Castle study site

Castle is the southernmost study site and administratively within Kirinyaga county. It is situated next to one of the mountain's major climbing routes "Kamweti", which starts

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from Castle Forest Lodge. It had one of the largest areas of eucalyptus plantation, created around 30 years ago, but it has since been harvested leaving only a small pocket of mature plantation. The site has an expansive continuous area of natural forest which, although protected, suffers from frequent illegal logging and harvesting of forest products. The entrance into the protected forest area, apart from a designated gate leading to Castle Forest lodge, is made possible through paths and entry points illegally created through the electric fence.

Chuka study site

The Chuka study site is in the southeastern side of Mount Kenya, within Tharaka nithi county. It is an area that had high population of *Ocotea usambarensis* but had been greatly impacted by unsustainable harvesting of indigenous trees in the 1990's. Exotic plantations cover an area of 200 ha but most of it has been harvested, leaving a small area of mature plantation. Main commercial tree species planted include Vitex (35%), Eucalyptus (50%), Grevillea (13%) and Pines (2%) (KFS, 2010). Other species in the Chuka plantation areas include mixed indigenous species like *Croton megalocarpus* and *Markhamia lutea*.

Ruthumbi study site

Ruthumbi forest is situated on the eastern slopes of Mount Kenya. It composes of extensive continuous natural forests and established (i.e., >20 years old) eucalyptus plantations. The Nyayo Tea Zone Development Corporation has planted tea crops on a 100 m strip between the forest edge, and community farms in some sections bordering forest in this study site. The rest of the study sites, the individual local community lands all extend up to the edge of the forest demarcated with electric fence (see Figure 3.4). The strip of tea alongside the forest in Ruthumbi site was planted to prevent the local community from encroaching on the forest land (Kinyua, 2002; KFS, 2010).

3.2.2 Forest types

Prior to carrying out the pilot study, a consultation was arranged and made with the respective forest station managers at each selected study site. This was specifically to seek their opinion regarding human-related disturbance intensities between the KFS/KWS

designated multiple-use zone (areas bordering the forest edges to 1 km into the forest interior), and the wilderness activity zone of the forests (i.e., areas beyond 1 km from the boundary into the interior) (Figure 3.5). After this, a consideration was arrived at regarding the multiple-use zone as a disturbed forest type and the wilderness activity zone area as an undisturbed forest type.

For the purposes of this study, undisturbed, disturbed and plantation forests are defined as described below:

(i) *Undisturbed*: The undisturbed forest area is part of the wilderness zone (section 3.1.4 (i)) and occurs 3-4 km from the forest edge. It is part of an extensive area with minimal human interference

(ii) *Disturbed*: Disturbed forest is part of the multiple-use zone which stretches as a 1 km belt into the protected forest area from the forest edge (section 3.1.4 (iv)). The multiple-use zone is frequently used by local communities to derive various benefits in the form of forest resources.

(iii) *Plantation*: Eucalyptus plantations established mainly in previously degraded sites or as afforestation where there were no trees before (section 3.1.4 (vi)).

3.3 Pilot study and sampling protocol

A pilot study was carried out between December 22nd, 2018, to February 24th, 2019, within the study sites of Chuka, Castle and Ruthumbi in the selected sections of undisturbed, disturbed and eucalyptus plantation forest types. The objectives of the pilot study were to test both the habitat and bird data collection protocols, determining the appropriate number of samples required at each study forest type, and to validate the classification of natural forest into undisturbed and disturbed forest by comparing selected habitat characteristics related to disturbance. To do these, a network of point count stations was established at each forest type within which bird data was collected, and where habitat characteristics data were collected within a 10 m radius circular plots, and also quadrats within the 10 m radius (see section 3.3.1 (b)). The point counts were placed in the disturbed, undisturbed and eucalyptus plantation forests along transects that were to yield 25 point count stations in each of the disturbed and undisturbed forest types in each study

site. In eucalyptus plantation, a maximum number of point counts that the plantation could accommodate were placed given their relatively small sizes in each study site. The first transect was placed at disturbed forest type considering accessibility (e.g., avoiding inaccessible terrain i.e., extreme steep valleys), and placing it to run perpendicular, 0-1 km, from the forest edge towards the forest interior (to coincide with the multiple use zone (section 3.1.4 (iv)). The first transect for the undisturbed forest was similarly selected based on accessibility but at between 3-4 km from the forest edge. Subsequent transects in both disturbed and undisturbed forest were each placed systematically at least 200 m from each other (Bibby et al., 1998), and with the number of transects that would generate 25 point counts in each forest type per site for the three sites, placed at least 200 m from each other, from forest edge, open glades, and from different vegetation type in a collective attempt to avoid any edge effects (Bibby et al., 1998; Wu et al., 2016). The use of 200 m interval from one transect to the other and from one point count station to the other was to avoid incidences of double counting of birds in forests, particularly those with large home ranges (Bibby et al., 2000; Volpato et al., 2009). This resulted in a combined 75 point count stations for each of the two forest types (undisturbed and disturbed) across all three study sites (Table 3.3).

Table 3.3: Distribution of point counts stations within each study site and forest type.

Forest type/site	Chuka	Castle	Ruthumbi	Total	Altitudinal range (m)
Undisturbed	25	25	25	75	1666-2576
Disturbed	25	25	25	75	1558-2255
Plantation	11	2	27	40	1551-2299
Total	**61**	**52**	**77**	**190**	

In eucalyptus plantations, the transects and respective point count stations were positioned as for both disturbed and undisturbed forests, ensuring that transects were at least 200 m from each other and point counts were also placed at least 200 m from each other and from the plantation edge, and from any major vegetation types that are not eucalyptus plantation (Figure 3.7). Eucalyptus plantation had a total of 40 point count stations, that was the maximum possible number given the size of each plantation at each study site and

to avoid non-independence (Table 3.3). The GPS location and altitude of each point count station was recorded. Figure 3.7 illustrates the placement of transects and point counts in each of the forest types in one of the study sites.
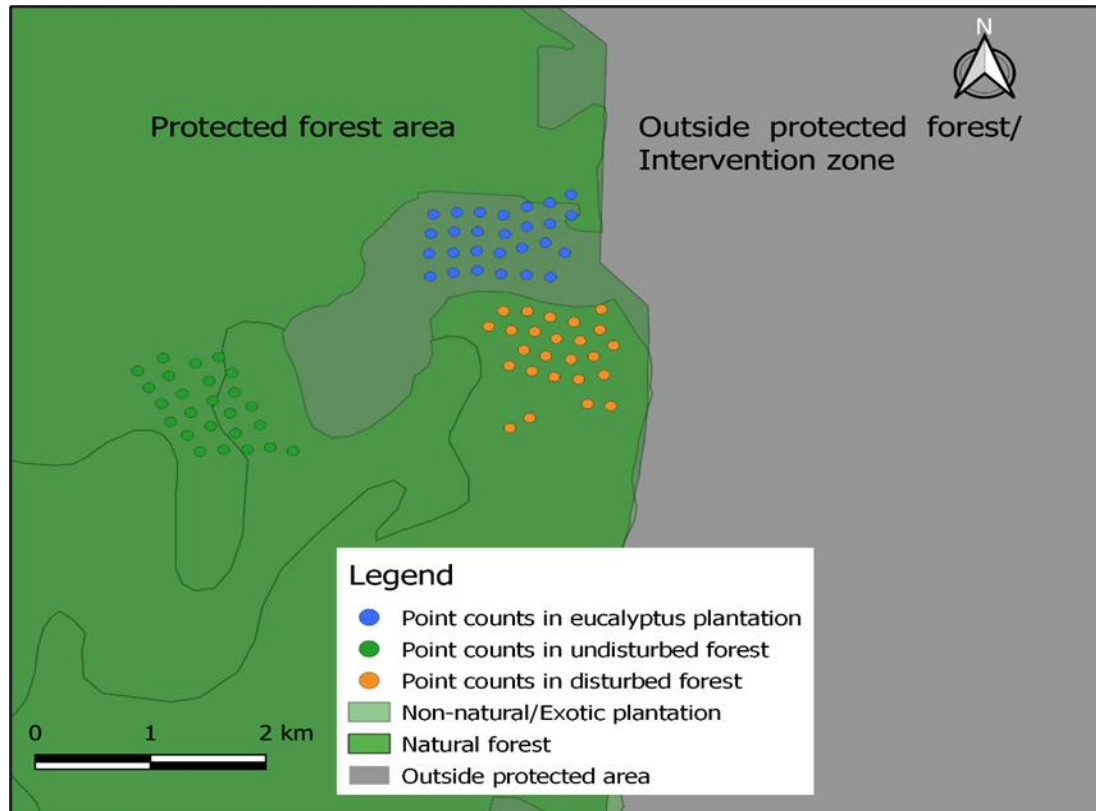
Figure 3.7: Map illustrating the point count station placement within the disturbed, undisturbed and eucalyptus plantation forest types in one of the study forests sites (Ruthumbi)**.**

All data collection was undertaken by both the researcher and a well-trained assistant who was experienced in bird and ecological data collection and habitat assessment within Mount Kenya. During the pilot, bird surveys were carried out between 6:00-10:00 am to coincide with peak bird activity (Sutherland et al., 2004). At each point count station, birds were identified for 10 minutes, using visual and acoustic recognitions, excluding birds flying over the point counts because it cannot be established where it originated from. The species of birds, number of individuals, and distances to the bird(s) from the center of the point count were recorded (Bibby et al., 2000; Buckland et al., 2001). The distance estimation to birds from center of point counts were perfected during pilot study,

when the researcher and research assistant practiced distance estimation to individual birds whether seen or heard by measuring the exact distances to spots where a bird was seen or heard calling/singing using a measuring tape. This was repeated for a number of time until the distance estimation was near perfect both in open and closed vegetations. Various distances to birds from centre of point counts were recorded, but only those that were within 50 m radius was considered for data analysis (see section 3.3.1).

Species accumulation curves were generated to determine the appropriateness of the sampling effort per forest type and found to be adequate as they were either close to or at asymptote (Figure 3.8). This strategy follows Ralph et al (1995), who recommended that the number of samples may be derived from the statistical evaluation of pilot data. It has also been suggested that in the absence of a pilot study, an absolute minimum of at least 30 point count stations should be established in a given habitat for it to be sufficiently sampled and to detect most of the biologically meaningful differences in a bird study, i.e. between different forest patch sizes or habitat types (Ralph et al., 1995; Bibby et al., 1998).
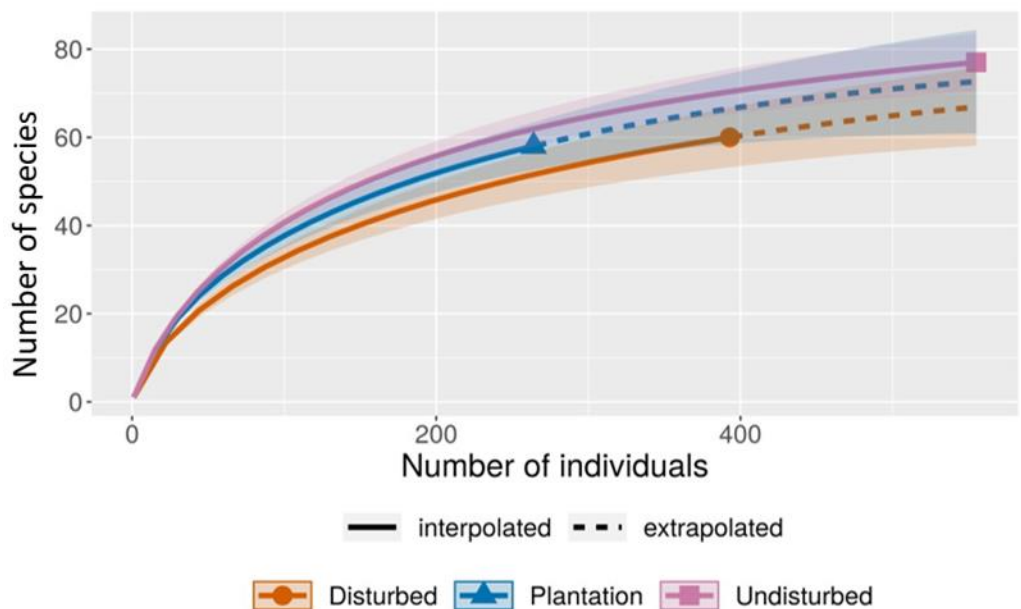
Figure 3.8: Species accumulation curves to determine appropriate sample size in disturbed (n = 75), undisturbed (n = 75) and eucalyptus plantation (n = 40) forest types.

During the pilot study, for the purpose of validation of wilderness zone and multiple use zone as undisturbed and disturbed forest types respectively, the habitat characteristics collected in the wilderness and multiple use zone were the number of cut trees and number of human trails/paths bisecting the 10 m radius of the point counts. Other habitat characteristics collected in trial basis and in practicing data collection methods during the pilot study in all forest types are described in detail in section 3.3.1. The definition of the habitat characteristics collected, and further justification is also detailed in Appendix 2.

Descriptive (i.e., means and standard deviations) and generalised linear models with Poisson distribution were used to validate the classification of the multiple-use zone and wilderness activity zone into disturbed and undisturbed forest types respectively for this study. Specifically, both the number of cut trees and trails were used as measures of disturbance and found to be significantly higher in the multiple-use zone (i.e., disturbed forest) than the wilderness activity zone (i.e., undisturbed forest) (Table 3.4). Following these therefore, the terms undisturbed and disturbed can be used to signify the difference in the two forest types. For instance, the less local human disturbance in undisturbed forest section and higher disturbance in disturbed forest section validates the definitions adopted in section 3.2.2 in this thesis.

Table 3.4: The differences in disturbance (number of cut trees and trails) between wilderness activity zone and multiple use zone using means and generalised linear model using log-link function. The values are counts of cut trees and human trails (mean ± SD) within a 10 m radius of the centre of each point count station. For each disturbance measure in each forest type, n = 75. LR = likelihood ratio, df = degree of freedom, p = statistical p-value.

Sites	Disturbance measures		Mean ± SD		Likelihood ratio test	
	No. of cut trees	No. of trails	No. of cut trees	No. of trails	Cut trees	Trails
Multiple use zone (0-1 km)	131	121	1.75 ± 1.35	1.61 ± 1.01	LR = 13.493, df = 1, p < 0.001	LR = 27.635, df = 1, p < 0.001
Wilderness activity zone (3-4 km)	20	42	0.27 ± 0.68	0.56 ± 0.68		

3.3.1 Data collection protocols and analysis

a) Bird data collection and classification

Bird data collected were the number of species per point count and number of individuals per species. These were to be used to determine bird species richness and relative abundance (hereafter abundance) at each forest type (relative abundance is used here since distance sampling was not undertaken to estimate population size (i.e., abundance)). The data were collected at each of the 190 point count stations established across the three study sites and three forest types (Table 3.3). Point-counts are considered a powerful method for measuring bird species richness, relative abundance, and density (Buckland et al., 2001) and particularly suitable for counting birds in spatially complex habitats (Bibby et al., 2000). Point counts have been widely used in bird surveys in tropical and subtropical forests (Rappole et al., 1998; Barlow et al., 2007 (a); Zou et al., 2012; Ocampo-Penuela and Pimms, 2015; Wu et al., 2016).

Main bird data collection using point counts were carried out for a year (April 2019-March 2020) across all four seasons, starting from the long rainy season (April-June), then short-dry season (July-September), short-rainy season (October-December) and long dry season (January-March). Point counts were carried out between 6:00 am to 10:30 am, which is the time when birds are most active and vocal in the tropics (Sutherland et al., 2004). They were also carried out only during fair weather conditions (i.e., no strong winds nor during heavy rains) as these can also affect the activeness and activities of birds and hence detectability. The researcher and the assistant approached a given point count and silently waited for 1 minute to allow any disturbed birds to settle and resume their normal activities. After this, data were collected for 10 minutes at each point count, before proceeding to the next point count separated by at least 200 m (Bibby et al., 1998; Bibby et al., 2000). Birds were identified using both sight and calls, and their identity, number of individuals and distances of birds from where they were first sighted/heard to the center of the point count were recorded (Bibby et al., 2000; Buckland et al., 2001). Birds recorded were allocated to distance bands (Bibby et al., 1998; Bibby et al., 2000) and only those within 50 m were used for analysis in this study, due to noticeable drops in

detectability beyond this distance (e.g., Schieck, 1997; Vergara et al., 2010). All birds flying overhead, although recorded, were not used during data analysis since it could not be determined where they originated from. Those flushed out within the 50 m radius and their origin determined within the 10-minute period of counting were recorded and included in analyses. Each point count station was sampled a total of eight times - twice during each of the four seasons. Consequently, a total of 1,520 point counts were conducted for this study. Repeat sampling within each season was separated by at least two weeks and for each repeat, the routes walked to survey each point count station was reversed from that for the first count. The reason for conducting point counts across each season was to capture all year-round bird presence and distribution, and to increase the chances of recording all the birds utilising the forest types in the lower parts of the Mount Kenya forest reserve. Taxonomic nomenclature for all birds followed Stevenson et al. (2004) and the latest version of the checklist of the Birds of Kenya, revised by the Bird Committee of East African Natural History of Society (Bird Committee, 2019).

Bird species were classified according to their forest dependency as either forest specialists (FS), forest generalists (FG), forest visitors (FV), or non-forest birds (NF), following Bennun et al. (1996). The FS and FG are forest birds which are dependent on forests, while FV and NF birds are not dependent upon forests. FS depend upon relatively intact, undisturbed forest, and are typical of the forest interior and are species that are most likely to disappear when the forest is greatly modified (Bennun et al., 1996). FG on the other hand may also occur in undisturbed forest but are able to exist - even becoming numerous - in modified forests. FV sometimes occur in forests but are more typical of other habitats, and since they are not dependent upon forests, they may survive in modified habitats even if all the forest disappeared. NF birds are birds that do not depend on forests and their presence in a forest may be an indication of intense forest disturbance.

Furthermore, birds were classified based on their dietary guilds (insectivores, frugivores, granivores, carnivores, nectarivores, and omnivores) by consulting Gray et al. (2007), Ndang'ang'a et al. (2013), Asefa et al. (2017), and Chiawo et al. (2018). Finally, bird species were also grouped into species of conservation concern. Specifically, those recognised as (i) Kenyan Mountain Endemic bird (KMEB) species (Evans and Fishpool,

2001), (ii) Afrotropical Highlands biome restricted species (ATHB) (Evans and Fishpool, 2001), and (iii) threatened under the International Union for Conservation of Nature (IUCN) Red List (IUCN, 2021).

The reason for classifying birds into forest dependency groups and dietary guilds was threefold: (i) they represent different ecological traits depicting important ecosystem functioning and, based on their ecological roles, can provide direct links to ecosystem processes rather than when considering all species as equivalent (e.g., Gagic et al., 2015); (ii) to complement the use of overall species richness, since, although it is widely used, it has been found not to be a suitable indicator of the impacts on local biodiversity. This is because species that go locally extinct due to ecosystem alteration can be replaced by others - often of lower conservation concern - with little or no impact on overall species richness (Dornelas et al., 2014; Hillebrand et al., 2018; Cazalis et al., 2020); (iii) species richness is a metric only related to taxonomic diversity (Naeem et al., 2012).

The reasons for including species of conservation concern, particularly those listed as either KMEB or ATHB species, is that they are likely to be highly sensitive to and threatened by modification and degradation of forest habitats, leading to possible reduction in their numbers and/or being replaced by widespread species and non-forest species (McKinney and Lockwood, 1999).

Prior to analysis, to avoid temporal pseudo-replication of bird data from the two-seasonal replicates, species richness was pooled within the season and maximum abundance per species was taken across the temporal replicates per point count station. Similarly, species richness was pooled across the four seasonal replicates. To obtain a single value of species abundance per point count station from the four seasonal replicates, the maximum abundance per species across the four seasons were used. This arrangement was made following the findings that seasons does not strongly predicts the key study responses, for instant, habitat characteristics in chapter 4, species richness and relative abundance in chapter 5 and species composition in chapter 6 as shown in Appendix 3 a and b.

b) Habitat characteristic justification and data collection

In order to determine habitat-bird relationships, at each point count station established for surveying birds, a complementary suite of habitat characteristics was also collected, broken down into measures of vegetation structure, human disturbance, altitude, slope and distance to forest edge. This data was collected within a radius of 10 m (approx. 0.04 ha) surrounding the centre of each point count station. A 10 m radius was used as this follows a method by James and Shuggart (1970) that has been regarded as an optimum plot size and shape for such studies (e.g., James and Shuggart, 1970; Krebs, 1989; McComb et al., 2010). It has also been used by other researchers in their studies of birds in tropical montane forests (e.g.,Werner et al., 2012; Zuluaga and Rodewald, 2015; Sisay et al., 2017). All habitat characteristic variables are defined and described in Appendix 2 and justified below.

Vegetation structure variables

It has previously been shown that tree height, canopy height and vegetation cover at various levels from the forest floor are important habitat features related to vegetation vertical structure (Rutten et al., 2015; Montgomery, 2001). Such measures have been widely found to have strong associations with bird species richness (Bibby et al., 2000; Huang et al., 2014). Canopy height, for example, has been shown to be an important indicator of forest biomass, and can influence species diversity, site quality, including being essential for studying forest micrometeorological phenomena and forest-atmosphere interaction (Tao et al., 2016). Tree diameter at breast height (DBH) is a measure of tree size and has also been found to positively correlate with bird species richness and diversity (Thinh, 2006). Tree size is useful in showing changes in tree size following selective logging and can be useful as a proxy to determine the impacts of human activities involving removal of trees in forests (Bibby et al., 2000; Peh et al., 2005).

Percentage canopy cover is defined here as the percentage of the ground covered by a vertical projection of the outermost perimeter of the natural spread of the foliage of trees (not considering the multiple layers) (Jennings et al., 1999). Percentage vegetation cover at different heights from forest floor is an important determinant of microclimates and microhabitats, and consequently determinants of the abundance and distribution of bird

species (MacArthur and MacArthur, 1961, Whittaker et al., 2001). However, intensities of anthropogenic disturbances affect foliage profiles at different heights resulting in a negative correlation between the amount of vegetation in the upper and lower strata (Bibby et al., 2000).

Litter depth is defined as the loose fallen vegetation especially twigs and leaves that cover the ground (Facelli and Pickett, 1991). Snags are dead, decaying but still standing trees. The importance of these variables in this study are that birds are exposed to litter depth and its contents (microbes and moisture), and it influences invertebrate presence (Willson and Comet, 1996) and hence distribution of birds, especially in the understory (e.g., Banks et al., 2017). Snags can easily form cavities and provide loose barks which can be used by birds to make nests (Vázquez and Renton, 2015), store food supplies and provide food sources (see section 2.5.1, Figure 2.2 for illustration photos taken during this study). Snags are also mainly targeted as a source of fuelwood since it is normally dry, easy to fell and to carry (e.g., Bate, 2008; personal observation in Mount Kenya forest reserve).

Basal area is defined as the sum of the cross-sectional surface of live trees, measured at DBH, and units are given per unit area (Balderas Torres and Lovett, 2013). Basal area is used in bird studies as a surrogate for overstory structure and has been reported to affects vegetation structure and determines habitat quality (Lee and Carroll, 2018). Tree distances from centre of point counts can be different in differently disturbed natural forests while in plantations can be influenced by silvicultural practices resulting to differently dispersed trees in different forest types. Silvicultural practices such as spacing during planting of plantation trees is common and affects forest management for birds (e.g., Finch and Stangel, 1993).

Number and density of saplings have been used to show the impacts of disturbance in tropical forests. For example, it has been used to show that disturbed forest has high potential for regeneration than undisturbed forest (Borah et al., 2014). The contribution of foliage height diversity on bird diversity is well known (Jayson and Mathew, 2003; Huang et al., 2014) and can be affected by anthropogenic disturbances and may differ between natural and exotic plantations.

Collection procedure: Tree heights and tree DBH were measured within point counts, with target trees selected using the point-centred quarter method (PCQ) (Mitchell, 2010). Each point count station was divided into four quadrants and the nearest tree identified ($\geq$ 7 cm DBH and at least 5 m height and not branched below 0.5m) in each quadrant and its height measured with the use of a Nikon forestry laser rangefinder-900 to the nearest 1 m. In cases where use of the rangefinder was impractical due to dense vegetation, visual estimation was used by a researcher with practical skills in visual estimation done prior to the study. Canopy heights at three levels (low $\leq$ 10m, mid 11-20 m and high > 20m) were similarly measured using a rangefinder and visual estimation. The DBH was then measured on the same trees that were measured for height with the use of the DBH meter held at breast height level (1.3 m from the base of the tree) and following recommendation by Dahdouh-Guebas and Koedam (2006). The DBH was also used in calculating trees basal area. Tree distance from the center of the point count for each tree measured for height and DBH was also determined with the use of a tape-measure. The use of the PCQ method is appropriate for the investigation of forest structure since it is not dependent on one forest type but can be widely applied and is simple and effective in characterising vegetation while minimising damage to the forest (Dahdouh-Guebas and Koedam, 2006).

Percentage canopy cover was measured using a Geographic Resource Solution (GRS) Densitometer by taking the reading at five points; i.e., center of point count, and at the four quadrants of the 10 m radius. The reason for multiple measurements was to increase measurement reliability. Vegetation cover profile at differing heights (0-1 m, 1-3 m, 3-5 m, 5-8 m, and > 8 m from forest floor), that denote the volume of vegetation at these heights in percentage coverage, was determined using visual estimation of each height category within a radius of 10 m around the center of each point count. These measurements were used to generate the Foliage Height Diversity (FHD) index using the Shannon-Weiner formula:

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

Where   Pi =proportion of all the individual foliage height category which belong to the ith category.

Litter depth was directly measured using a measuring ruler at five location/points within each 10 m radius point count circle i.e., one point at the center of point count, and one point at the center of each of the four quadrants of the circle. The number of standing snags (dead, decaying and standing trees ($\geq$7 cm DBH and $\geq$1 m height)) were directly counted within each of the whole 10 m radius circle. Number of saplings (growing tree seedlings of $\geq$ 30 cm height, and < 3 cm dbh) (Dendy et al., 2015) were counted within a 1 m x 1 m quadrat placed at the centre of the circle and at the centre of each of the four quadrants.

Human disturbance variables

This includes the number of fallen trees, including the dead rotting fallen trees/logs. Number of cut trees is defined here as any vascular vegetation from small and larger trees that has been freshly cut (with signs of recent cuttings- i.e., fresh cuts) with a tool, and with signs of cutting in the roots, trunks, or branches. The fallen/downed trees and logs can be used by plants to take root (i.e., act as nurse logs). In addition, fallen/falling trees open the canopy, influencing light penetration and therefore can influence the undergrowth characteristics (Lawton, 1990) that are likely to influence birds. Number of cut trees can show the extent of the forest cutting and is a sign of anthropogenic disturbance (Zhou et al., 2013), associated with the removal of habitat components that would have been available for bird use. Small-scale local disturbance occasioned by cuttings have been identified as a significant process in all major forest biomes (Forsman et al., 2013). The presence and number of paths/trails on the other hand, can signify the presence of people in forests that can disturb wildlife i.e., perceived as potential predators by wildlife (Bötsch et al., 2018). Human activities rely on trails, which intersect an otherwise contiguous habitat (Bötsch et al., 2018). Bibby et al. (2000) noted that number and width of paths may be useful in characterising human disturbance in forest.

Collection procedures: Number of fallen/downed trees was counted within the circle formed by the 10 m radius of each point count station. Number of cut vegetation, i.e., small trees and larger trees that have been freshly cut (from subjective visual inspection and determination of fresh/recently cut versus aged/old cuts) and with signs of cutting in the roots, trunks and branches were physically counted within the circle. Old cuts were

not counted as it does not show recent disturbance. Number of human trails/paths intersecting the circle were also counted.

Slope, altitude, and distance from forest edge

Altitude is expressed as the height asl and is a determinant of bird distribution (Campos-Cerqueira et al., 2017; Ghimire et al., 2021). For example, bird species may move to higher elevation following warming climates (Forero-Medina et al., 2011). Slope on the other hand is useful in forest bird studies since it has been noted to determine forest vegetation structure (Pascal and Pelissier, 1996), and possibly indirectly affects birds. Distance from forest edges is important in bird studies as an indicator of general forest quality (e.g., Gehlhausen et al., 2000), nesting success (Flaspohler et al., 2001) and is a correlate of disturbance and hunting (Lee and Marsden, 2008).

Collection procedure: Altitude was measured using a hand-held GPS (Garmin GPSMAP 64S) that automatically records the altitude in meters to an accuracy between 10 to 20 m. Slope was determined by obtaining a mean gradient from two measurements taken from the center of each point count to one end of the 10 m radius. Two 1 m long rods were each placed 10 m apart upslope and downslope (center of point count to upslope/downslope end), and using a clinometer, the slope angle was measured and averaged. Distance to the forest edge was the perpendicular distance from each point count station to the forest edge, measured using the handheld GPS in meters. This was easily done from the edge of the forest.

3.3.2 Data Analysis

Details of the relevant statistical analyses are included in the methodology sections of each data chapter. All statistical analyses were performed in SPSS version 26 or Minitab version 19. The use of other specific software/package for statistical analysis and visualisation are detailed in specific chapters where applicable in this thesis.

For the statistical analysis to be performed appropriately, the data must meet a variety of assumptions specific to the test (Mishra et al., 2019). All analyses that utilised a continuous response variable were checked for normality using Shapiro-wilk's test and

consulting Q-Q plots and histograms. Furthermore, the size and direction of skewness was checked for all continuous variables for independent tests of differences with two or more groups (Ghasemi and Zahediasl, 2012). For non-normal continuous variables, data were transformed using either square-root or log transformation depending on the amount and direction of skewness (e.g., Tabachnick and Fidell, 2007) before testing using ANOVA. Non-normal proportional data on the other hand were transformed using logit transformation (Warton and Hui, 2011).

Count data were not transformed prior to analysis in Chapters 4 and 5 but instead analysed using Generalised linear models (GLM) with a loglinear function which is recommended over log or square-root transformation of these data (O'Hara and Kotze, 2010). But first the count data were checked if it fulfills the assumptions required to validate the use of the model. The use of GLMs applied to count data has become popular because it has the advantage of yielding directly interpretable estimates on an additive scale and can be used except when a decision against a fixed type 1 error rate is more important than estimating a parameter on its original scale (St. Pierre et al., 2018). ANOVA test was used for continuous and non-binomial proportional data because it is a robust test, even for small violations of normality (Scheff, 2016). Kruskall-wallis test is a non-parametric test that was used for greatly skewed non-normal continuous data. Specific tests of other assumptions are described in the methods section of subsequent data chapters.

3.3.3. Ethics and permissions

Ethical approval was granted by the Animal Welfare and Ethical Review Board of the University of Brighton on 23rd August 2018. The permit to do research in Kenya was granted on 17th October 2018 by the National Commission for Technology and Innovation (NACOSTI) (Permit No. NACOSTI/P/18/17034/25292), while permission to access and do research in Mount Kenya forest reserve was given by Kenya Forest Service (Permit No. RESEA/1/KFS/VOLIII(154)). Kenya Wildlife Service granted permission to access Mount Kenya National Park and its environs on 25th October 2018 (Permit No. KWS/BRM/5001).

Chapter 4 - Habitat characteristics in undisturbed, disturbed and eucalyptus plantation forest types.

4.1 Introduction

Undisturbed forests in tropical areas have been deemed irreplaceable owing to its contribution to tropical forest biodiversity conservation (Gibson et al., 2011). This is through its ability to maintain greater biodiversity than found in disturbed forests, whose suitability have been reduced through, for example, anthropogenic actions such as forest conversion to agriculture or logging (Barlow et al., 2007 (a); Barlow et al., 2016). Unfortunately, undisturbed forests are now few and fast reducing both in size and distribution in the tropics (Kobayashi, 2001; UNEP and FAO, 2020). This is especially the case within the tropical montane forests, where the existing forests are fragmented and degraded, leaving isolated remnants across the landscape (Adhikari et al., 2020). At the same time, disturbed and exotic plantation forests are rapidly expanding (Bremer and Farley, 2010; Gibson et al., 2011), resulting in a continuum of forest types with, potential, considerable differences in forest structure and habitat characteristics. These differences in habitat structure and characteristics may depend on factors such as nature of forest (native or exotic) and degree of forest degradation through disturbance (Adhikari et al., 2020; Bentsi-Enchill et al., 2022).

Habitat characteristics demonstrate how complex or simple the forest structure and habitat is and determine the presence/absence of resources and conditions for organisms' survival and fitness (Seidler, 2017; Mwasapi and Rija, 2021). Forest habitat characteristics are dictated by forest vegetation features related to both horizontal and vertical structural arrangement of trees and their branches (Kimmins, 2004; Rutten et al., 2015). It may describe variability of plant assemblages at a local scale and reflect characteristics of the ecosystem's complexity (Adhikari et al., 2020). Since habitat characteristics are related to the structural complexity of ecosystems, and provide habitat quality for fauna (Rutten et al., 2015), they have been used at local scale level to model species diversity (Erdelen, 1984; Huang et al., 2014). For instance, characteristics such as forest canopy heights have been found to be indicators of forest biomass and site quality (Li et al., 2018), while tree

DBH shows changes in tree size parameters, and has been used as a proxy to determine impact of human activities that involve removal of trees in forests (Peh et al., 2005).

But anthropogenic disturbance potentially destabilises and disintegrates the forest structure complexity. For instance, unprecedented anthropogenic actions form the main contributor to loss of undisturbed forest and emergence of modified habitats (disturbed and plantation forests) in the tropical montane forests (Morris, 2010; Bentsi-Enchill et al., 2022). This is following the need by humans to often obtain and fulfill their socio-economic and cultural requirements from forests (Jenkins and Schaab, 2018; Euler et al., 2012), and in the process degrade the forests. Emergence and expansion of exotic plantations in tropical montane forests are due to the need to quickly restore cleared and degraded forest sites. The preference for exotic plantations for this purpose is based on their rapid growth and suitability for commercial timber, and their potential role in biodiversity conservation. The consequence of these anthropogenic actions is altered forest structures, with degree of alteration dependent upon the intensity of human induced pressures (Malhi et al., 2014). Disturbed forests sites are likely to have different habitat quality from undisturbed forest sites, reducing their ability to sustain biodiversity. Similarly, exotic plantations present habitat characteristics that are similar or different from those existing in natural forests, potentially complementing natural forest in supporting forest biodiversity. The extent of these emerging forest characteristic differences or similarities among emerging forest types, present conservation, and management challenges in tropical montane forests. Conservationists and managers may need to understand the dynamics of forest structure, as influenced by anthropogenic actions, to formulate appropriate management and conservation strategies and actions in tropical montane forests.

More specifically in the tropical montane forests, anthropogenic actions that mainly affect forests arise from forest dependent people living within a 5 km distance from the forest boundary (Wass, 1995; Popradit et al., 2015; Hishe et al., 2021; Beche et al., 2022). The ensuing forest degrading actions from these people arise from physical access to the forests, formation, and use of trails in the forests (e.g., Figure 4.1 a), removal of vegetation

through cutting (e.g., Figure 4.1 b), and breaking, uprooting, or cutting and general creation of disturbed points within the forests (Figure 4.1 c).
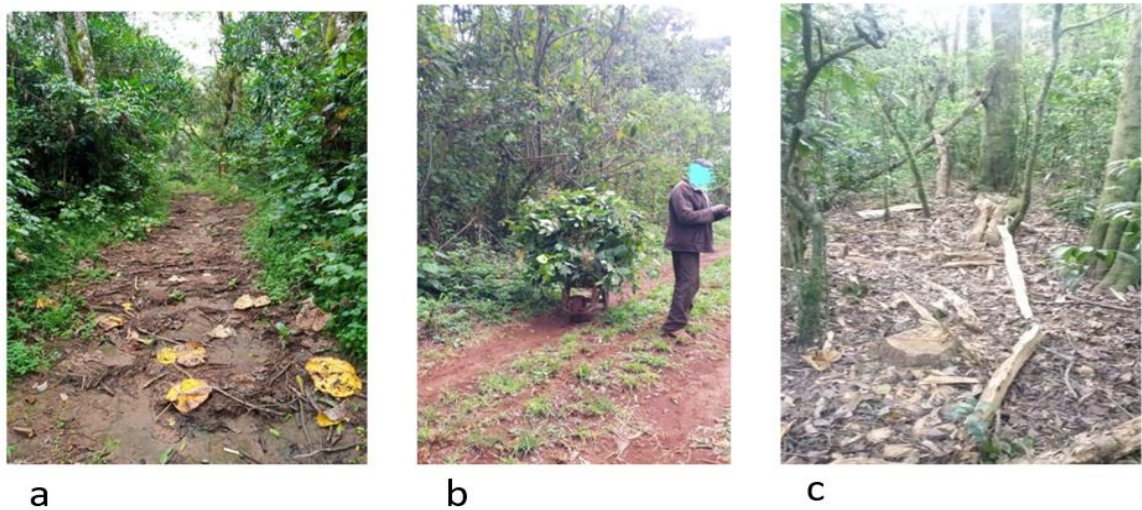
Figure 4.1: (a) human paths/trails cutting through the forest in Mount Kenya forest reserve. Photo taken on 24/06/2019 (b) Cut or uprooted leaves and branches from forest as livestock feed from disturbed sites of Mount Kenya forest reserve. Photo taken on 14/10/2019 (c) site with cut and felled trees and general site disturbance following selective logging in the disturbed sites of Mount Kenya forest reserve. Photo taken on 16/10/2019. Photo credit: Author.

The anthropogenic actions (Figure 4.1 a-c) potentially interfere directly with forest vegetation structural arrangements and indirectly may affect the biodiversity dependent on these habitat characteristics (Opuni-Frimpong et al., 2021). The actions may lead to disruption of the provision of essential resources and conditions for other taxa depending on these characteristics (e.g., MacArthur and MacArthur, 1961; Garden et al., 2007). For example, human use of forests has been shown to interfere with forests' capacity to maintain their original biodiversity (Groombridge and Jenkins, 2000). When the vegetative parts of the plant are removed, the microhabitat conditions of the local area is negatively affected; for instance, it can result in changes to the amount of light energy and photoperiod, humidity and temperature within the forest (Cochard, 2011; Nakamura et al., 2017; Pincebourde and Salle, 2020), leading to interference of site productivity. At the same time, the cover against predators is removed thus exposing certain organisms to more predation (May and Norton, 1996; Eggers et al., 2008), while affecting prey availability for others (May and Norton, 1996; Dorresteijn et al., 2015). This can influence

the occupancy and the viability of the organisms that inhabit the disturbed environment (Eggers et al., 2008; Fryxell et al., 2020) and so it may affect their diversity. Studies have also indicated that more anthropogenic impacts are expected along the forest edges and to about 1 km into the forest interior (e.g., Haddad et al., 2015; Beche et al., 2022).

The consequence of anthropogenic disturbances has been shown in tropical forests of Thailand and Indonesia to reduce species diversity of woody plants near forest boundaries (Riswan and Hartanti, 1995; Popradit et al., 2015), and to associate with reduced tree and seedling species richness, density, and biomass in Ethiopian sacred church forests (Cardelús et al., 2019). In Borneo, Malaysia, evidence of significant changes in forest canopy structure and foliage traits along forest edges was attributed to local anthropogenic disturbance, including reduced canopy structure and plant traits related to light capture and growth (Ordway and Asner, 2020). In a study in tropical forests of Bhuban hills in India, the average tree density and basal area were significantly higher in undisturbed forests, with poor regeneration status in disturbed forest, all attributed to anthropogenic disturbance (Borah et al., 2014).

Despite these studies, there are limited understandings of habitat characteristics that characterise different forest types, resulting from different intensities of anthropogenic disturbance actions in an otherwise continuous forest, as well as those of exotic plantations in tropical montane forest. Past studies have observed that there is far less attention given to the role of degree of anthropogenic modification of forests used to extract wood and non-wood products in Afrotropical forests (Erb et al., 2018; Grantham et al., 2020), and how this changes forests' structural characteristics (Seidler, 2017). This is despite Afrotropical forests experiencing illegal forest disturbance, as well as existence of anthropogenic actions on forests supported by forest management policies, such as the local community involvement and benefits through Participatory Forest Management (PFM) approaches (Heinrich, 1997; Wily, 2002). Through PFM approach, community members are permitted to access and extract resources such as honey, grazing materials, fuelwood and for other socio-cultural uses and recreational purposes. These potentially cause habitat structural change, and thus lead to existence of different forest types (i.e.,

disturbed forest type in accessed and utilised areas, and undisturbed forests type in un-accessed areas).

The extent and degree of change of habitat characteristics across local anthropogenically caused natural forest types have not been assessed in Afrotropical montane forests. In addition, the habitat characteristics of the current widespread expansion of exotic plantation within protected forest in the tropics (Wormald, 1992) have not been determined, yet its roles in biodiversity conservation is contested (e.g., Brockerhoff et al., 2008; Thijs et al., 2014). Also, the exotic plantations' structural characteristics relative to natural disturbed and undisturbed forest is unknown, especially the extent to which it resembles or differs with natural forests. Furthermore, there is need to determine if indeed, the classification of forest types adopted as undisturbed, disturbed and plantation is supported by habitat characteristics at each forest type.

At best, few studies (e.g., Hitimana et al., 2004; Wethered and Lawes, 2005; Waltert et al., 2005; Cronin et al., 2014; Asefa et al., 2017; Gebeyehu et al., 2019) have looked at the change in forest structures resulting from anthropogenic disturbance in Afrotropical montane forests, and none have assessed characteristics of introduced plantations relative to natural forests. Consequently, little scientific information is available for comparative studies, and to guide forest management decision making and management strategies in Afrotropical montane forests, and particularly where there is integration of community participation in forest management. Therefore, this study aims to determine (1) the habitat characteristics variation and similarities within and across three forest types (undisturbed, disturbed and eucalyptus plantations) in Mount Kenya forest reserve (2) if variation of habitat characteristics represents classification of forest types into undisturbed, disturbed and plantation. This is achieved through assessment of variation of habitat characteristics within and across forest types.

4.2 Methods

4.2.1 Study design and data collection

The study site, study design and data collection procedure for all habitat characteristic variables (vegetation structure, human disturbance, and slope, altitude, and distance from forest edge) used in the study and presented in this chapter are detailed in Chapter 3. These data were collected either for four consecutive seasons, starting from the long-rainy season (April - June 2019) to the long-dry season (January-March, 2020) (for data that can significantly change seasonally; i.e. dynamic variables) or collected once during the entire data collection period (for those variables that do not change significantly seasonally; i.e. static variables) (See Table 4.1).

Table 4.1: Habitat characteristic variable groupings into vegetation, human disturbance and slope, altitude, and distance from forest edge. Data categories and frequency of collection with its justification are also shown.

Variables grouping	Data type categories	Frequency of data collection	Justification
Vegetation structure variables			
Tree heights	Continuous	Once	Static
Tree DBH	Continuous	Once	Static
Basal area	Continuous	Once	Static
Tree distance	Continuous	Once	Static
Low canopy height at 0-10 m	Continuous	Once	Static
Mid canopy height at 11-20 m	Continuous	Once	Static
High canopy height at > 20 m	Continuous	Once	Static
Litter depth	Continuous	Seasonal	Dynamic
Foliage height diversity	Continuous	Seasonal	Dynamic
Percentage vegetation cover at 0-1 m	Proportional	Seasonal	Dynamic
Percentage vegetation cover at 1-3 m	Proportional	Seasonal	Dynamic
Percentage vegetation cover at 3-5 m	Proportional	Seasonal	Dynamic
Percentage vegetation cover at 5-8 m	Proportional	Seasonal	Dynamic
Percentage vegetation cover at > 8 m	Proportional	Seasonal	Dynamic
Percentage canopy cover	Proportional	Seasonal	Dynamic
Number of saplings	Count	Once	Static
Number of snags	Count	Once	Static
Human disturbance variables			
Number of cut trees	Count	Seasonal	Dynamic
Number of trails	Count	Seasonal	Dynamic
Number of fallen trees	Count	Seasonal	Dynamic
Slope	Continuous	Once	Static
Altitude	Continuous	Once	Static
Distance from the forest boundary	Continuous	Once	Static

For this study, seasonal difference did not strongly predict the habitat characteristics (see appendix 3 a) hence an average value of the seasonal replicates for the dynamic variables were used per point count since the aim here was not to investigate for seasonal variation. Taking an average also avoided temporal pseudoreplication and an artificially inflated sample size. Therefore, there were a total of 190 point counts separated into disturbed forest (75-point counts), undisturbed forest (75-point counts) and eucalyptus plantation (40-point counts) that were used to collect these variables and hence these make the total sample size (see Chapter 3 for details).

4.2.2 Data analysis

a) Descriptive analysis.

Habitat characteristics were first explored via generation of descriptive statistics such as means, standard error, minimum, median, maximum and range (Appendix 4) within each forest type across sites. This was aimed at summarising the main characteristics of the variables.

b) Statistical analysis.

To investigate habitat characteristic variation within forest types across sites, one-way ANOVA or generalised linear model (GLM) tests were used. For these, habitat characteristics were selected in turn as response variables while forest type was placed as a grouping predictor factor. Assumption of homogeneity of variance for ANOVA-based analysis was checked using Levene's test. When the assumption of homogeneity of variance was not met or in cases of unequal sample sizes, Welch's ANOVA test was used to determine significance of variation rather than F-test in ANOVA (Tomarken and Serlin, 1986). This is because Welch's ANOVA is robust and helps to protect against committing type 1 error even when sample sizes are unequal. Post-hoc sources of observed differences were shown based on Tukey's test for ANOVA and Dunn's test for Kruskal-wallis.

To assess the habitat variables that are characteristic of each forest type (i.e., habitat variables that contribute most to the differences), percentage indicator values of each variable was calculated across forest types using indicator species analysis (IndVal) in

PAST version 4.02 (Hammer, 2001). The statistical significances (p values ≤ 0.05) of the indicator values were estimated by 9999 random reassignments (permutations) of sites across groups (Hammer, 2001).

The habitat variables were further investigated for multi-collinearity as detailed in Chapter 3 section 3.3.2. This was aimed as an initial step to remove non-important collinear variable data from subsequent multivariate analysis. Principal Component Analysis (PCA) was then performed on the remaining habitat characteristic variables to further reduce the number of variables to obtain the number that can be subjected to further analysis. Before subjecting data for PCA, the many data variables (more than 12 variables) were split into two groups of variables to facilitate the generation of PCs that can be effectively interpreted. The variables that were closely related such as those describing tree sizes, foliage cover etc were used as criteria for splitting these variables and aimed at reducing the number of variables in a way that it can be interpreted with ecological meaningful themes. Kaiser Meyer Olkin (KMO) and Bartlett's test of sphericity were used to check the appropriateness of data for PCA and to determine if indeed there was a need to perform data reduction using PCA. The acceptable KMO value range is from 0.5 to 1 (Hutcheson and Sofroniou, 1999; Field, 2000; Pallant, 2013) while that of Bartlett's test of sphericity, the significant value less than 0.05 is acceptable. To name the generated principal components after extraction based on their factor loadings, and to correctly interpret them, a rotation of factor approach was adopted (Pallant, 2013) where an oblique rotation was used (Oblamin method of rotation with Kaiser Normalization). This was chosen because in reality, there is a certain amount of correlation between forest vegetation and environmental variable factors (Bradfield and Scagel,1984; Li et al., 2020), thus relying only on the outcomes of orthogonal rotation (i.e., varimax rotation- one of the methods of orthogonal rotation) can lead to the loss of valuable information if there is a correlation between factors (e.g., Costello and Osborne, 2005). Castello and Osborne (2005) further justified the use of oblique rotation by saying that it creates a correct and similar pattern of loadings and recommend its use because it presents a more realistic simple statistical structure. The generated component scores from PCA were then subjected to one-way ANOVA, Welch's ANOVA, Kruskal-Wallis, and T-test depending on their applicability to explore variation in habitat related components within forest types

and variation across forest types. Post-hoc sources of observed differences were based on Tukey's test for ANOVA and Welch's ANOVA and Dunn's test for Kruskal-Wallis.

4.3 Results

4.3.1 Habitat characteristics within each forest types across study sites

a) Descriptive statistics for habitat characteristic variables

Descriptive statistics for the 23 habitat characteristics within forest types across sites (pooled sites) are summarised in Appendix 4.

i) Habitat characteristics with highest mean values in each forest type

Table 4.2: Habitat characteristics with significantly different means across forest types across sites (p < 0.05 tested using ANOVA or GLM), but with (a) highest mean (mean ± SE, n=75) in undisturbed forest type (shaded green) (b) highest mean (n=75) in disturbed forest type (shaded orange) (c) highest mean (n=40) in eucalyptus plantation (shaded blue). DBH = diameter at breast height, PCS = point count station., SE = standard error.

Habitat characteristics	Undisturbed forest	Disturbed forest	Plantation forest
Tree DBH (cm)	91.52 ± 4.81	63.46 ±3.48	68.31 ± 4.04
Tree basal (m²)	0.79 ± 0.08	0.39 ± 0.04	0.42 ± 0.04
% Vegetation cover at 5-8 m height	31.94 ± 1.25	29.64 ± 0.9	15.34 ± 1.15
% Vegetation cover at > 8 m height	26.05 ± 1.44	21.15 ± 0.86	12.02 ± 0.87
% Canopy cover	80.22 ± 0.68	71.57 ± 2.01	46.79 ± 3.05
Litter depth (cm)	2.49 ± 0.09	2.23 ± 0.07	2.09 ± 0.10
Number of snags (counts)	0.98 ± 0.09	0.73 ± 0.09	0 ± 0.00
Distance from forest edge (m)	3767.10 ± 41.80	710.2 ± 37.4	739.4 ± 64.2
Altitude (m)	2098.00 ± 38.00	1915.9 ± 27.5	2032.6 ± 47.4
% Vegetation cover at 3-5 m	29.85 ± 1.56	34.71 ± 1.35	20.02 ± 1.84
Foliage height diversity (H' index))	1.45 ± 0.01	1.49 ± 0.01	1.34 ± 0.03
Number of saplings (counts)	6.63 ± 0.60	8.85 ± 0.47	0.38 ± 0.12
Number of fallen trees (counts)	1.27 ± 0.11	1.32±0.10	0.3 ± 0.08
Tree height (m)	14.73 ± 0.59	13.16 ± 0.45	17.45 ± 0.94
Tree distance from center of PCS (m)	4.00 ± 0.14	3.10 ± 0.13	4.38 ± 0.22
% Vegetation cover at 0-1 m	40.44 ± 2.8	44.91 ± 2.08	60.8 ± 4.01
Number of cut trees (counts)	0.47 ± 0.10	1.89 ± 0.15	2.08 ± 0.18
Number of trails (counts)	0.67 ± 0.09	1.61 ± 0.11	1.88 ± 0.19

With consideration of habitat characteristics that had significant differences between forest types across sites only, undisturbed forest type (n = 75), had nine habitat characteristic variables with highest mean relative to disturbed (n = 75) and plantation (n = 40) forests as shown in Table 4.2 (shaded green). Disturbed forest had four habitat characteristics with highest means relative to other forest types (shaded orange), while eucalyptus plantation had five habitat characteristics with highest means (blue shade) as compared to disturbed and undisturbed forest type (Table 4.2).

ii) Habitat characteristics with lowest mean values in each forest type

Undisturbed forest type across sites had the lowest significantly different measures of means of the following: percentage vegetation cover at 0-1 m, number of cut trees and number of trails (Table 4.2). Disturbed forest type had lowest mean measurement of tree DBH, tree basal area, distance of point counts from forest boundary, tree heights, distance of trees from centre of each point count station, and altitude. Eucalyptus plantation had lowest mean measures of percentage vegetation covers at 3-5 m, 5-8 m, and > 8 m, and canopy cover, litter depth, FHD and number of saplings (Table 4.2).

b) Habitat characteristics variation

i) Across forest types (pooled study sites)

When all the study sites are pooled together, habitat characteristics across forest types show that a total of 18 habitat variables (13 vegetation structure, 3 human disturbance, altitude, and distance from forest edge variables) were all significantly different across forest types, across all the sites (Table 4.3).

Table 4.3: Significant (p < 0.05) and non-significant (p ≥ 0.05) habitat characteristic variables (vegetation structure, disturbance, slope, altitude, and distance from forest edge) across forest types. (n = 190).

Variable	Test	Test statistic	p-value
Vegetation structure variables			
Tree height	One-way ANOVA	F = 10.04	< 0.001
Tree DBH	One-way ANOVA	F = 13.61	< 0.001
Basal area	One-way ANOVA	F = 17.47	< 0.001
Tree distance	One-way ANOVA	F = 12.02	< 0.001
Percentage vegetation cover at 0-1 m	Kruskal-wallis	H = 19.21	< 0.001
Percentage vegetation cover at 1-3 m	One-way ANOVA	F = 1.80	0.168
Percentage vegetation cover at 3-5 m	One-way ANOVA	F = 18.27	< 0.001
Percentage vegetation cover at 5-8 m	One-way ANOVA	F = 47.51	< 0.001
Percentage vegetation cover at > 8 m	One-way ANOVA	F = 28.50	< 0.001
Percentage canopy cover	One-way ANOVA	F = 62.36	< 0.001
Low canopy height at 0-10 m	Kruskal-wallis	H = 0.71	0.700
Mid canopy height at 11-20 m	Kruskal-wallis	H = 1.44	0.488
High canopy height at > 20 m	Kruskal-wallis	H = 1.06	0.588
Foliage height diversity	Kruskal-wallis	H = 30.22	< 0.001
Litter depth	One-way ANOVA	F = 6.06	0.003
Number of saplings	GLM	χ2 Likelihood ratio = 459.08	< 0.001
Number of snags	GLM	χ2 Likelihood ratio = 62.11	< 0.001
Human disturbance variables			
Number of cut trees	GLM	χ2 Likelihood ratio = 55.90	< 0.001
Number of trails	GLM	χ2 Likelihood ratio = 36.89	< 0.001
Number of fallen trees	GLM	χ2 Likelihood ratio = 45.24	< 0.001
Slope	One-way ANOVA	F = 2.86	0.060
Altitude	Kruskal-wallis	H = 14.34	0.001
Distance from the forest edge	One-way ANOVA	F = 1650.83	< 0.001

ii)   Within each forest type (across study sites)

Undisturbed forest

Of the 23 habitat characteristics assessed, 18 (78 %) varied significantly within undisturbed forest type across study sites (Table 4.4). This includes 14 vegetation structure variables, 2 human disturbance variables, slope, and altitude.

Table 4.4: Significant and non-significant variation in habitat characteristics within undisturbed forest types across study sites (n = 75).

Variables	Test	Test statistic	p-value
Vegetation structure variables			
Tree height	One-way ANOVA	F = 2.01	0.141
Tree DBH	One-way ANOVA	F = 6.26	< 0.001
Basal area	One-way ANOVA	F = 6.79	0.002
Tree distance	One-way ANOVA	F = 5.27	0.007
Percentage vegetation cover at 0-1 m	Kruskal-wallis	H = 49.37	< 0.001
Percentage vegetation cover at 1-3 m	Welch's ANOVA	Welch = 36.45	< 0.001
Percentage vegetation cover at 3-5 m	One-way ANOVA	F = 15.285	< 0.001
Percentage vegetation cover at 5-8 m	Welch's ANOVA	Welch = 21.03	< 0.001
Percentage vegetation cover at > 8 m	Welch's ANOVA	Welch = 28.21	< 0.001
Percentage canopy cover	One-way ANOVA	F = 1.74	0.183
Low canopy height at 0-10 m	Kruskal-wallis	H = 0.77	0.679
Mid canopy height at 11-20 m	Kruskal-wallis	H = 11.54	0.003
High canopy height at > 20 m	Kruskal-wallis	H = 8.74	0.013
Foliage height diversity	Kruskal-wallis	H = 19.37	< 0.001
Litter depth	Welch's ANOVA	Welch = 33.64	< 0.001
Number of saplings	GLM	χ^2 Likelihood ratio = 220.84	< 0.001
Number of snags	GLM	χ^2 Likelihood ratio = 7.92	0.019
Human disturbance variables			
Number of cut trees	GLM	χ^2 Likelihood ratio = 1.22	0.544
Number of trails	GLM	χ^2 Likelihood ratio = 8.49	0.014
Number of fallen trees	GLM	χ^2 Likelihood ratio = 9.66	0.009
Slope	One-way ANOVA	F = 3.41	0.038
Altitude	Kruskal-wallis	H = 65.80	< 0.001
Distance from the forest edge	One-way ANOVA	F = 0.75	0.478

Disturbed forest

Only 10 of the 23 habitat characteristics assessed varied significantly within disturbed forest type across sites. This includes 8 vegetation structure variables, slope, and altitude. No human disturbance varables were significantly different (Table 4.5).

Table 4.5: Significant and non-significant variation in habitat characteristics within disturbed forest types across study sites (n = 75).

Variables	Test	Test statistic	p-value
Vegetation structure variables			
Tree height	Welch's ANOVA	Welch = 1.88	0.164
Tree DBH	One-way ANOVA	F = 3.26	0.044
Basal area	One-way ANOVA	F = 2.79	0.069
Tree distance	One-way ANOVA	F = 1.79	0.174
Percentage vegetation cover at 0-1 m	Kruskal-wallis	H = 2.22	0.330
Percentage vegetation cover at 1-3 m	Welch's ANOVA	Welch = 15.84	< 0.001
Percentage vegetation cover at 3-5 m	Welch's ANOVA	Welch = 17.52	< 0.001
Percentage vegetation cover at 5-8 m	One-way ANOVA	F = 17.03	< 0.001
Percentage vegetation cover at > 8 m	Welch's ANOVA	Welch = 11.97	< 0.001
Percentage canopy cover	One-way ANOVA	F = 2.72	0.073
Low canopy height at 0-10 m	Kruskal-wallis	H = 3.18	0.220
Mid canopy height at 11-20 m	Kruskal-wallis	H = 9.34	0.009
High canopy height at > 20 m	Kruskal-wallis	H = 1.27	0.530
Foliage height diversity	Kruskal-wallis	H = 1.35	0.510
Litter depth	One-way ANOVA	F = 9.24	< 0.001
Number of saplings	GLM	$\chi 2$ Likelihood ratio = 16.12	< 0.001
Number of snags	GLM	$\chi 2$ Likelihood ratio = 5.64	0.060
Human disturbance variables			
Number of cut trees	GLM	$\chi 2$ Likelihood ratio = 2.16	0.339
Number of trails	GLM	$\chi 2$ Likelihood ratio = 0.46	0.794
Number of fallen trees	GLM	$\chi 2$ Likelihood ratio = 0.39	0.823
Slope	One-way ANOVA	F = 6.52	0.002
Altitude	Kruskal-wallis	H = 65.79	< 0.001
Distance from the forest edge	One-way ANOVA	F = 1.87	0.161

Eucalyptus plantation

Only six habitat characteristic variables differed significantly within the plantation across sites. These included percentage vegetation cover at 0-1 m, 1-3 m, and 3-5 m from forest floor, high canopy height at > 20 m, altitude, and number of fallen trees (Table 4.6).

Table 4.6: Significant and non-significant variation in habitat characteristics within plantation forest types across study sites (n = 40).

Variables	Test	Test statistic	p-value
Vegetation variables			
Tree height	One-way ANOVA	F = 3.05	0.059
Tree DBH	One-way ANOVA	F = 0.57	0.571
Basal area	One-way ANOVA	F = 0.72	0.492
Tree distance	One-way ANOVA	F = 0.23	0.799
Percentage vegetation cover at 0-1 m	Kruskal-wallis	H = 7.86	0.020
Percentage vegetation cover at 1-3 m	One-way ANOVA	F = 18.93	< 0.001
Percentage vegetation cover at 3-5 m	One-way ANOVA	F = 6.68	0.003
Percentage vegetation cover at 5-8 m	One-way ANOVA	F = 0.72	0.495
Percentage vegetation cover at > 8 m	One-way ANOVA	F = 0.61	0.548
Percentage canopy cover	One-way ANOVA	F = 1.69	0.199
Low canopy height at 0-10 m	Kruskal-wallis	H = 1.33	0.513
Mid canopy height at 11-20 m	Kruskal-wallis	H = 0.26	0.879
High canopy height at > 20 m	Kruskal-wallis	H = 6.71	0.035
Foliage height diversity	Kruskal-wallis	H = 0.69	0.709
Litter depth	Welch's ANOVA	Welch = 13.73	0.074
Number of saplings	GLM	$\chi 2$ Likelihood ratio = 1.68	0.196
Number of snags*			
Human disturbance variables			
Number of cut trees	GLM	$\chi 2$ Likelihood ratio = 0.24	0.889
Number of trails	GLM	$\chi 2$ Likelihood ratio = 1.85	0.396
Number of fallen trees	GLM	$\chi 2$ Likelihood ratio = 3.85	0.050
Slope	One-way ANOVA	F = 0.38	0.685
Altitude	Kruskal-wallis	H = 26.21	< 0.000
Distance from the forest boundary	One-way ANOVA	F = 2.10	0.137

*Number of snags were not recorded within the plantation forest

4.3.2 Habitat variables that are characteristic to each forest type.

Based on percentage indicator values (% IndVal), Table 4.7 shows variables with the highest indicator value assigned to each forest type.

Table 4.7: Habitat characteristic variables showing the highest percentage indicator values (% IndVal) ranging from 0-100 % for habitat variables assigned for each forest types: undisturbed forest with indicator values ranging from 35-72 %, disturbed forest with indicator values ranging from 35-55 % and plantation forest types with indicator values ranging from 38-42 %. The highest significant ($p < 0.05$) values indicate a very good indicators of habitat variables that are characteristics to a given forest type (and composed of A and B components, A indicating specificity *or* positive predictive value of the habitat characteristic as indicator of the forest type while B is the second conditional probability – fidelity or sensitivity of the habitat characteristic as indicator of the target forest type). Eight (8) habitat characteristics therefore were characteristics to undisturbed forest types (all associated with undisturbed forest conditions), six (6) were characteristics to disturbed forest while five (5) were characteristics to eucalyptus plantation.

Habitat characteristic variables	% IndVal	p-value
Undisturbed forest type		
Tree DBH (cm)	40.98	< 0.001
% Vegetation cover at 5-8 m	41.42	< 0.001
% Vegetation cover at > 8 m	43.99	< 0.001
% Canopy cover	40.40	< 0.001
Low canopy height at 0-10 m	33.53	0.3736
Litter depth (cm)	36.50	< 0.001
Number of snags (counts)	43.01	< 0.001
Altitude (m)	34.70	0.0038
Distance of point counts from forest edge (m)	72.21	< 0.001
Disturbed forest type		
Slope (degrees)	37.68	0.034
% Vegetation cover at 1-3 m	35.64	0.0951
% Vegetation cover at 3-5 m	41.04	< 0.0001
Canopy height at 11-20 m	34.05	0.0677
Canopy height at > 20 m	32.09	0.0369
Foliage height diversity (FHD) (H' index)	34.89	< 0.001
Number of saplings	55.84	< 0.001
Number of fallen trees	35.87	< 0.001
Plantation forest types		
Tree height (m)	38.37	< 0.001
Tree distance from center of point counts (m)	38.12	< 0.001
% Vegetation cover at 0-1m	41.6	< 0.001
Number of cut trees (counts)	42.11	< 0.001
Number of trails (counts)	38.36	< 0.001

4.3.3 Variable reduction using multicollinearity analysis and Principal Component Analysis (PCA).

a) Multicollinearity

The results from multicollinearity analysis, as the initial step of variable reduction using Pearson correlation coefficients and variance inflation factor (VIF) values, revealed that tree DBH and tree basal area were collinear (Appendix 5 a). The two variables (Tree DBH and Tree basal area) were highly correlated (Pearson correlation = 0.956, $p < 0.001$, n = 190) (Appendix 5 b) and both showed VIF values of above 10, indicating that there is multicollinearity with these two variables (i.e., are not independent of each other). However, when either of tree DBH and tree basal area is removed, both the VIF values becomes less than 10, indicating that multicollinearity is removed (Appendix 5 a). Since basal area is a derivative of tree DBH, the basal area was therefore removed from subsequent analysis.

b) PCA

Twenty-two habitat characteristic variables remained after tree basal area was removed, and these were divided into two groups of variables for PCA analysis. This resulted in a group of 10 variables that relate more to tree heights, distances from forest edge, altitude, and slope (Table 4.8) while the other 12 relate to proportion of vegetation cover at various heights, disturbance, litter depth and saplings (Table 4.9). Out of the first 10 habitat characteristic variables analysed, four principal components (PCs) were derived accounting for a total of 62 % of the total variation in the original data (Table 4.8), while the second set also resulted to four PCs accounting for 73 % of the total variation in the original data (Table 4.9). The choice of four PCs was based on the Kaiser's criteria or the use of an Eigenvalue cut-off point of above 1. This is where any variance less than 1 is considered to contain less information than one of the original variables and so is not worth retaining (Jolliffe, 2002).

Table 4.8: The first group of PCs retained with eigenvalues above 1, with 62 % of total variation explained.

	Initial Eigenvalues			Rotation Sums of Squared Loadings
PCs	Total	% of Variance	Cumulative %	Total
1	2.053	20.528	20.528	1.880
2	1.698	16.984	37.512	1.551
3	1.474	14.738	52.250	1.491
4	1.011	10.113	62.363	1.511
5	0.918	9.176	71.539	
6	0.833	8.331	79.870	
7	0.634	6.337	86.206	
8	0.551	5.508	91.714	
9	0.465	4.652	96.366	
10	0.363	3.634	100.000	

Table 4.9: The second group of PCs retained with eigenvalues of above 1, with 73 % of total variation explained.

	Initial Eigenvalues			Rotation Sums of Squared Loadings
PCs	Total	% of Variance	Cumulative %	Total
1	3.684	30.699	30.699	3.310
2	2.353	19.608	50.307	2.286
3	1.497	12.478	62.785	1.721
4	1.173	9.773	72.558	2.200
5	0.713	5.939	78.496	
6	0.542	4.518	83.015	
7	0.518	4.313	87.328	
8	0.471	3.922	91.250	
9	0.422	3.515	94.765	
10	0.312	2.600	97.365	
11	0.217	1.805	99.170	
12	0.100	0.830	100.000	

In the habitat characteristics data considered, the minimum requirement for Kaiser Meyer Olkin (KMO) were met in both groups of data (i.e., KMO = 0.558 and 0.687 respectively),

showing that the data were adequate and there is confidence in performing PCA. The Bartlett's test requirement was also met with significant values of $p < 0.05$ (i.e., Approx. chi-square = 268.18, df = 45, $p < 0.001$ and Approx. chi-square = 1013.80, df = 66, $p < 0.001$ respectively) indicating that the observed correlation matrix was significantly different, inferring that a correlation matrix for the measured variables is significantly different from an identity matrix, hence consistent with the assumption that the matrix is factorable. It also shows that the data do not produce an identity matrix and are thus approximately normal and acceptable for further analysis (Pallant, 2013; Field, 2000).

The four principal components (PC1 to PC4 in each of the two groups, Table 4.10) were then named by using factor loading in each component generated by interpreting the pattern matrix in oblique rotation method. The factor loading with a minimum value of 0.4 was used and coefficient display was sorted by size (see Table 4.10).

Table 4.10: Pattern matrix for habitat characteristic variables and derived PCs. The bold figures are major factor loadings in each PC. The loadings form related habitat characteristics.

Pattern Matrix				
	Component			
Habitat characteristic variables	**PC1**	**PC2**	**PC3**	**PC4**
First group				
Tree distance from centre of point count (m)	**0.772**	-0.215	-0.002	-0.109
Tree height (m)	**0.769**	0.129	-0.146	0.012
Tree diameter at breast height (DBH) (cm)	**0.691**	0.138	0.265	0.255
Canopy height at 11-20 m (Mid canopy)	-0.091	**0.845**	0.040	0.274
Canopy height > 20 m (High canopy)	0.237	**0.622**	0.012	-0.417
Canopy height at 0-10 m (Low canopy)	-0.018	**0.527**	-0.013	-0.112
Distance from forest boundary (m)	0.182	-0.069	**0.799**	0.222
Number of snags (counts)	-0.134	0.107	**0.770**	-0.168
Altitude (m)	0.023	-0.106	0.176	**0.821**
Slope (degrees)	-0.116	-0.076	0.369	**-0.543**
Second group				
Percent vegetation cover at 5-8m	**0.872**	0.006	0.008	-0.060
Percent vegetation cover at > 8m	**0.740**	-0.271	-0.168	0.068
Foliage height diversity (FHD)	**0.656**	0.206	0.318	-0.288
Litter depth (cm)	**0.622**	-0.257	-0.190	0.371
Percent Canopy Cover	**0.599**	0.212	-0.258	-0.170
Percent vegetation cover at 1-3 m	-0.100	**0.944**	-0.057	0.039
Percent vegetation cover at 3-5 m	0.409	**0.746**	0.125	-0.216
Percentage vegetation cover at 0-1 m	-0.389	**0.683**	-0.143	0.350
Number of cut trees (counts)	-0.123	-0.147	**0.843**	-0.068
Number of trails (counts)	-0.011	0.096	**0.772**	0.310
Number of fallen trees (counts)	-0.130	-0.033	-0.186	**-0.842**
Number of saplings (counts)	0.209	-0.023	0.033	**-0.742**

The naming of the components based on the highest factor loadings in each PC was done by identifying their ecologically meaningful common theme and labelling the factor based on that common theme (Table 4.11).

Table 4.11: Names of the components based on their ecologically meaningful common themes as indicated by factor loadings.

	Factor loading
First group	
PC1 Tree size	
Tree diameter at breast height (DBH)	0.691
Tree height (m)	0.769
Tree distance from centre of point count (m)	0.772
PC2 Low to high canopy height	
Low canopy height at 0-10 m	0.527
Mid canopy height at 11-20 m	0.845
High canopy height at > 20 m	0.622
PC3 Proximity to boundary and snags	
Number of snags	0.770
Distance from forest boundary (m)	0.799
PC4 Slope and altitude	
Slope (degrees)	-0.543
Altitude (m)	0.821
Second group	
PC1 Litter, high vegetation cover and diversity factors	
Percent Canopy Cover	0.535
Foliage height diversity (FHD)	0.590
Litter depth (cm)	0.684
Percent vegetation cover at > 8m	0.802
Percent vegetation cover at 5-8m	0.861
PC2 Low to mid vegetation cover factors	
Percent vegetation cover at 1-3 m	0.874
Percent vegetation cover at 3-5m	0.870
PC3 Forest disturbance factors	
Number of trails	0.767
Number of cut trees	0.881
PC4 Forest regeneration and tree fall	
Number of saplings	-0.754
Number of fallen trees	-0.896

Each PC had associated scores which were generated and treated as new variables. These new variables were used to assess statistical variation in habitat characteristics within each

of the forest types and across the forest types. The variables were treated as response variables with study sites and forest types as grouping variables.

4.3.4 Variation of the derived habitat characteristic components across forest types and within each forest type

a) Variation across forest types for pooled study sites

All except one (low, mid, and high canopy heights) of the derived habitat characteristics components, differed significantly across forest types, across sites (Table 4.12).

Table 4.12: Derived habitat characteristic components showing test results for components across forest types (pooled study sites) (n = 190). Post-hoc sources of observed differences are shown based on Tukey's test for ANOVA and Dunn-Bonferroni test for Kruskal-Wallis.

Component metric	Test	Test statistics	Df	p-value	Source of difference		
					Forest type(i) - Forest type(j)	i-j±SE	p-value
Tree size	ANOVA	F= 18.84	2	< 0.001	Undisturbed-Disturbed	0.79±0.14	< 0.001
					Plantation-Disturbed	0.90±0.18	< 0.001
					Plantation-Undisturbed	0.11±0.18	0.811
Low, mid & high canopy height	Kruskal-Wallis	H= 2.42	2	0.298	N/A	N/A	N/A
Proximity to boundary and existence of snags	Kruskal-Wallis	H= 129.04	2	< 0.001	Plantation-Disturbed	40.83±10.77	< 0.001
					Plantation-Undisturbed	114.03±10.7	< 0.001
					Disturbed-Undisturbed	-73.20±8.98	< 0.001
Slope and altitude	ANOVA	F= 8.15	2	< 0.001	Undisturbed-Disturbed	0.62±0.16	< 0.001
					Undisturbed-Plantation	0.16±0.19	0.686
					Plantation-Disturbed	0.46±0.19	0.040
Litter depth, canopy cover and foliage diversity factors	Kruskal-Wallis	H= 63.80	2	< 0.001	Plantation-Disturbed	65.78±10.77	< 0.001
					Plantation-Undisturbed	84.90±10.77	< 0.001
					Disturbed-Undisturbed	-19.12±8.98	0.100
Low to mid vegetation cover	Kruskal-Wallis	H= 14.54	2	< 0.001	Plantation-Undisturbed	21.59±10.77	0.135
					Plantation-Disturbed	40.55±10.77	< 0.001
					Undisturbed-Disturbed	18.96±8.98	0.104
Forest disturbance factors	Kruskal-Wallis	H= 78.50	2	< 0.001	Undisturbed-Disturbed	70.77±8.98	< 0.001
					Undisturbed-Plantation	-5.00±10.77	< 0.001
					Disturbed-Plantation	-4.23±10.77	0.100
Forest regeneration and fallen trees	ANOVA	F= 48.52	2	< 0.001	Undisturbed-Disturbed	0.28±0.13	0.088
					Plantation-Disturbed	1.54±0.16	< 0.001
					Plantation-Undisturbed	1.25±0.16	< 0.001

b) Variation within each forest type across sites.

Within the undisturbed, disturbed and eucalyptus plantation forest types across the study sites, each showed different habitat characteristics variation: Undisturbed forest type had six components being significantly different across sites (Table 4.13).

Table 4.13: Habitat characteristics within undisturbed, disturbed and plantation forest type across sites. N = 75 for undisturbed and disturbed forest and n = 38 for plantation forest (for plantation forest type, Castle was not included in the analysis due to its small sample size of n = 2). Post-hoc sources of observed differences are shown based on Tukey's test for ANOVA and Dunn's test for Kruskal-wallis. The mean differences are positive differences of forest type (i) and forest type (j) with associated p-values.

Components	Test	Test-statistics	df	p-value	Source of difference Forest site (i) – Forest site (j)	i-j ± SE	p-value
Undisturbed forest type							
Tree size	ANOVA	F= 5.14	2	0.008	Chuka-Castle	0.10±0.25	0.913
					Ruthumbi-Chuka	0.64±0.25	0.034
					Ruthumbi-Castle	0.74±0.25	0.012
Low, mid & high canopy height	Welch's ANOVA	Welch = 6.74	2	0.003	Chuka-Ruthumbi	0.56±0.25	0.080
					Castle-Chuka	0.41±0.25	0.249
					Castle-Ruthumbi	0.97±0.26	0.001
Proximity to boundary and existence of snags	ANOVA	F = 2.783	2	0.069	N/A	N/A	N/A
Slope and altitude	ANOVA	F = 120.08	2	< 0.001	Castle-Chuka	1.32±0.15	< 0.001
					Ruthumbi-Chuka	2.38±0.15	< 0.001
					Ruthumbi-Castle	1.06±0.15	< 0.001
Litter depth, canopy cover and foliage diversity factors	Welch's ANOVA	Welch = 53.96	2	< 0.001	Chuka-Castle	1.24±0.21	< 0.001
					Chuka-Ruthumbi	1.35±0.21	< 0.001
					Castle-Ruthumbi	0.12±0.21	0.842
Low & mid vegetation cover	Welch's ANOVA	Welch = 35.48	2	< 0.001	Castle-Chuka	1.63±0.22	< 0.001
					Castle-Ruthumbi	0.74±0.22	0.004
					Ruthumbi-Chuka	0.89±0.22	< 0.001
Forest disturbance factors	Kruskal-Wallis	H = 1.72	2	0.424	N/A	N/A	N/A
Forest regeneration and tree fall	ANOVA	F = 38.15	2	< 0.001	Chuka-Castle	0.93±0.19	< 0.001
					Ruthumbi-Chuka	0.70±0.19	0.001
					Ruthumbi- Castle	1.64±0.19	< 0.001
Disturbed forest type							
Tree size	ANOVA	F = 2.589	2	0.082	N/A	N/A	N/A
Low, mid & high canopy height	ANOVA	F = 0.361	2	0.698	N/A	N/A	N/A

Components	Test	Test-statistics	df	p-value	Source of difference Forest site (i) – Forest site (j)	i-j ± SE	p-value
Proximity to boundary and existence of snags	ANOVA	Welch = 35.45	2	0.067	N/A	N/A	N/A
Slope and altitude	ANOVA	F = 57.31	2	< 0.001	Castle-Chuka	1.32±0.15	< 0.001
					Ruthumbi-Chuka	1.46±0.15	< 0.001
					Ruthumbi-Castle	0.15±0.15	0.592
Litter depth, high vegetation cover and diversity factors	Welch's ANOVA	Welch = 11.74	2	< 0.001	Chuka-Castle	0.71±0.15	< 0.001
					Chuka-Ruthumbi	0.76±0.15	< 0.001
					Castle-Ruthumbi	0.51±0.15	0.942
Low & mid vegetation cover	Welch's ANOVA	Welch = 23.86	2	< 0.001	Chuka-Ruthumbi	0.31±0.21	0.306
					Castle-Chuka	0.78±0.21	< 0.001
					Castle-Ruthumbi	1.10±0.21	< 0.001
Forest disturbance factors	Welch's ANOVA	Welch = 2.35	2	0.107	N/A	N/A	N/A
Forest regeneration and tree fall	ANOVA	F = 3.71	2	0.066	N/A	N/A	N/A
Plantation forest type							
Tree size	T-test	T = 1.43	36	0.163	N/A	N/A	N/A
Low, mid & high canopy height	T-test	T = 1.81	36	0.079	N/A	N/A	N/A
Proximity to boundary and existence of snags	T-test	T = - 3.93	36	< 0.001	-0.31±0.08	N/A	N/A
Slope and altitude	T-test	T = 7.42	36	< 0.001	-1.43±0.19	N/A	N/A
Litter depth, high vegetation cover and diversity factors	T-test	T = -0.05	36	0.965	N/A	N/A	N/A
Low & mid vegetation cover	T-test	T = 5.08	36	< 0.001	1.44±0.28	N/A	N/A
Forest disturbance factors	T-test	T = 0.16	36	0.873	N/A	N/A	N/A
Forest regeneration and tree fall	T-test	T = - 1.95	36	0.059	N/A	N/A	N/A

Figure 4.2 presents a summary of derived habitat characteristic components that differed significantly and those that did not within each forest type.
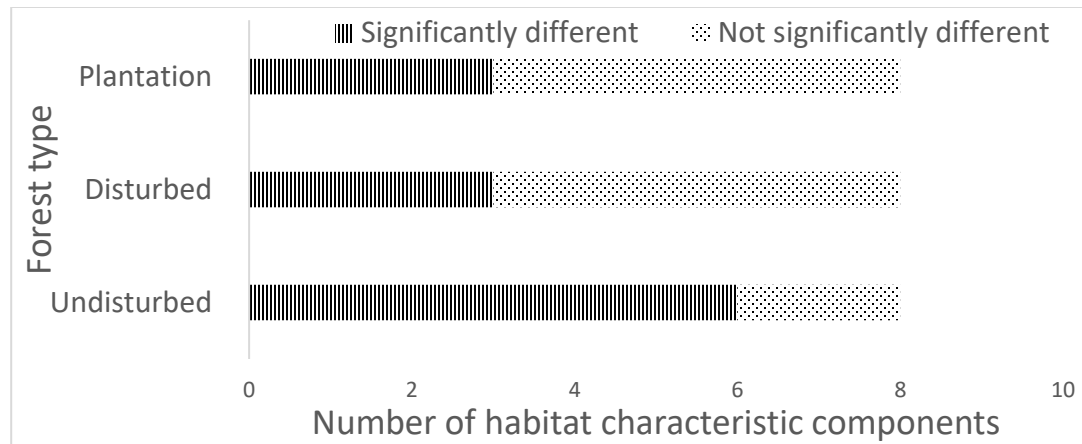
Figure 4.2: Representation of number of habitat characteristic components that differed significantly and those that did not differ significantly within undisturbed, disturbed and eucalyptus plantation forest types across sites. Six components differed significantly within undisturbed forest type while only three components differed significantly within disturbed and eucalyptus plantation respectively across sites.

4.4 Discussion

4.4.1 Variation in habitat characteristics in undisturbed, disturbed and eucalyptus plantation forest types

In this study, the undisturbed forest type across sites was indeed characterised by habitat features depicting relatively pristine or less disturbed forests of the tropics. Undisturbed forest type had higher number of vegetation structural characteristics with high mean measurements compared to disturbed and plantation forest types. In addition, undisturbed forest was found to be significantly characterised by high indicator values of between 35-72 % of habitat characteristic mainly related to vegetation structural arrangements (Table 4.7). The characteristics included trees size represented by tree DBH, high vegetation covers from mid to high and overall canopy cover, deep litter depth and high number of snags. These characteristics, except for altitude and distance from forest edge, all relate to vegetation structures that depicts and portray forest structural complexity (Caviedes and Ibarra, 2017). These are characteristics typical of undisturbed and mature forest in tropical forest (Tews et al., 2004; Ndang'ang'a, et al., 2013; Casas et al., 2016), although they are also reported in temperate areas (e.g. Whelan and Maina, 2005; Goetz et al., 2007; Caviedes and Ibarra, 2017). The relatively larger size of trees and denser vegetation

covers in the undisturbed forest type in this study is attributed to less human accessibility and use of the interior of the forest. Undisturbed forest type was located at a mean distance from the forest edge of 3767.10 ± 41.80 m (Table 4.2, Appendix 4) and possibly this had deterred both illegal loggers or permitted human access to collect fuelwood and other forest products. This also explain why there is higher mean number of snags in undisturbed forest type, where it is difficult for firewood collectors to reach because of distance and remove dead standing trees for fuelwood, unlike in easily accessed or disturbed forest areas. Similar findings have been reported elsewhere e.g., reduction of tree size with increasing level of disturbance in lowland tropical rainforest of eastern Himalaya (Gogoi and Sahoo, 2018; Naidu and Kumar, 2016).

The 18 out of 23 habitat characteristics (Table 4.3), and 7 out of 8 PCA derived habitat components (Table 4.12) that varied significantly across forest types shows that all forest types considered in this study are varied and are likely deriving their inherent differences based on these characteristics (mainly structural in nature) within each forest type. The differences is brought about by exotic nature of eucalyptus plantation as one of the forest types and anthropogenic disturbance is likely to be responsible for the differences observed in natural forests (disturbed and undisturbed forest types). Anthropogenic activities within the natural forests have mainly been reported to be responsible for changing forest structural arrangements (e.g., Evariste et al., 2010; Popradit et al., 2015). The anthropogenic disturbance within natural forest can therefore lead to differences in habitat structural characteristics in disturbed parts of the forest, compared to undisturbed forest.

Further assessment of habitat characteristics variables within each forest type revealed a progressive decrease in habitat characteristics differences (variability) from undisturbed forest, to disturbed and eucalyptus plantations. For instance, undisturbed forest had 18 out of 23 (78 %) habitat characteristics variables varying significantly within it (Table 4.4), disturbed forest type had 10 out of 23 or 43 % (Table 4.5) and eucalyptus plantation forest had 6 out of 23 or 26 % (Table 4.6). Similarly, the PCA derived habitat characteristic components that differed significantly reduced from undisturbed to eucalyptus plantation (Table 4.13). These shows that undisturbed forest type varied greatly structurally,

followed by disturbed forest type and then eucalyptus plantation that showed more structural similarities (i.e., more structurally homogenous). It is possible that human impacts in disturbed forest have reduced habitat structural complexity while eucalyptus plantation intrinsically have little variation as trees are all the same age and size and undergo similar silvicultural maintenance. Watson et al. (2004) similarly found a lot of variability of vegetation variables within undisturbed lowland tropical forest in Madagascar. These findings reveal that, apart from nature of eucalyptus plantation being more structurally homogenous, local anthropogenic disturbance are potentially capable of changing forest structural characteristics leading to a forest type which is structurally less heterogenous than undisturbed one.

Significant percentage indicator values shows that disturbed forest type is characterised by vegetation cover between 3-5 m from forest floor, canopy heights greater than 20 m, foliage height diversity (FHD), high number of saplings and number of fallen trees (Table 4.7). This shows disturbed forest has high shrub level vegetation, probably owing to more open forest cover that encourages understory bushes and shrubby vegetation (e.g., Mestre et al., 2017), including a high probability of tree sapling establishment (e.g., Wilder et al., 1999; Sterck et al., 2003). High FHD in the disturbed forest type is probably due to differential modification of the existing high percentage of understory vegetations, and foliage arrangement and structure by anthropogenic disturbance activities. The activities could include for example, cutting of livestock feeds, cutting where to place beehives, and firewood collections leading to increases in foliage diversity as also implied in Camprodon and Brotons (2006) and Leaver et al. (2019). Unlike in undisturbed forest where the vegetation structure is more complex due to absence of disturbance, anthropogenic disturbance could be contributing to promoting occurrence of a variety of foliage-based characteristics and conditions at shrub level hence likely greater habitat diversity and niches (see Leaver et al., 2019). Disturbance could also trigger and favor the abundant growth of many opportunistic shrubs, while more open conditions promote more tree falls due to wind (Arriaga, 2000) with falling trees causing falls of other trees (e.g., van der Meer and Bongers, 1996). From the current study, removal of trees with low canopy heights at 0-10 m, possibly by human related disturbance, may only allow mid and high canopy height trees to occur at the disturbed forest type. In addition, selective

logging, and removal of fuelwood (preference made on dead and dry trees) and harvesting of livestock feed (see Figure 4.1 (b) and (c)) could be responsible for less vegetation cover near the forest floor in disturbed forest. Anthropogenic disturbance is also probably the cause of generally more homogeneous forest type based on the recorded small number of habitat characteristic variables that differed significantly within the disturbed forest type (Table 4.5 and Table 4.13) as compared to undisturbed forest (Table 4.4 and Table 4.13). Disturbed forest also occurred at a distance closer to the forest edge (i.e., located at a mean distance of 710.2 m from forest boundary) than undisturbed forest (Table 4.2, Appendix 4) and possibly easily accessed by people and therefore experiencing intense anthropogenic disturbance. These findings agrees with that of Hager and Dohrenbusch (2011), that frequent disturbance leads to low canopy height and low basal area in tropical montane forests in Costa Rica. It also agrees with that of Gogoi and Sahoo (2018), that shrub density, and herb density significantly increased with increasing disturbance level in lowland tropical rainforest of eastern Himalaya, India. Jackson et al. (2006) also argued that the intensity of canopy and basal area removal and the degree of ground disturbance, are factors responsible for altering forest understory diversity and composition. Watson et al. (2004) reported that in southeastern Madagascar tropical forest, the edge habitats had significantly higher amounts of low and medium shrub cover, a less complete canopy, and less litter cover than the core habitats. Unlike in this study, Gogoi and Sahoo (2018) also recorded high sapling density largely in undisturbed sites. The high number of saplings in the disturbed forest type in the current study in relation to undisturbed forests could be due to good seedling germination and regeneration capacity in the largely open canopy sites but poor full establishment to trees owing to disturbance incidence (e.g., Gebeyehu et al., 2019).

In the study area, the plantation forests of eucalyptus have been established with the aim of provision of commercial wood and timber. In addition, they are also aimed to restore degraded forests and to offer opportunities for biodiversity conservation (Lindenmayer et al., 2003) within the tropical montane forests. But the specific contribution of the plantation species to ecological integrity in terms of habitat characteristics in relation to natural forests in montane forest areas is less clear. The current study has demonstrated that, relative to the natural forest types (undisturbed and disturbed), eucalyptus plantation

forest is characterised by higher significant indicator values for tree heights, tree distance from centre of point counts (i.e., trees are further apart and hence less dense), percentage of vegetation cover at 0-1 m, number of cut trees and number of trails (Table 4.2). Typically, plantations such as eucalyptus can be characterised by exceptional structural features, given that their growth characteristics are modified by man through selective and silvicultural processes (Sein and Mitlöhner, 2011). The resultant habitat characteristics structural arrangement features can probably introduce additional or unique features, that contribute to addition niches for organisms to strive in Afromontane forests. The introduced features by eucalyptus plantations can be complementary to that in the natural forests (undisturbed and disturbed). Tree heights for example, were found to be significantly taller within the eucalyptus plantation than any other forest type. This can be argued that eucalypts, by their nature and following management interventions (i.e., silvicultural activities) are taller, and this may create niches for some organisms to use. For instance, tall trees can be utilised for hunting by birds of prey (as illustrated in Figure 4.3).

Figure 4.3: Black/Great sparrowhawk (*Accipiter melanoleucus*) perching (possibly) hunting prey at Ruthumbi eucalyptus plantation. Photo taken on 27/04/2019. Photo credit: Author.

Vegetation cover from mid to higher heights (3 to >8 m), and overall canopy cover, showed the lowest values in the eucalyptus plantation, which possibly favors certain organisms that require open vegetation as their habitats (Table 4.2). More openings within

eucalyptus plantation may favor establishment of more or denser near ground vegetation cover, and possibly more light tolerant shrubs. It is not surprising therefore that apart from tall trees and high disturbance, eucalyptus plantation was significantly characterised by denser near ground vegetation at between 0-1 m (Table 4.7). It is worth noting that vegetation cover at between 0-1 m and 1-3 m in the eucalyptus plantation was mostly native vegetation that regenerated as undergrowth, possibly some through seeds originating from natural forests nearby, and some invasive species such as *Lantana camara* (personal observation). Disturbance represented by cut trees and trails are expected in eucalyptus plantation owing to high human activities and expected silvicultural works on eucalyptus. However, the low number of fallen trees is probably because eucalyptus plantations are managed and not prone to windfalls nor incidental felling due to their growth characteristics, unlike in natural forests. Lack of snags in plantation forests in the current study is expected due to the silvicultural practices. It is probably also because eucalyptus trees are meant to be harvested after several years for timber, and hence do not attain natural death and form snags.

Based on habitat characteristic variation within eucalyptus plantation across sites (Table 4.6) (including derived habitat characteristic components (Table 4.13)) eucalyptus plantation is more homogenous than any other forest type. It had only 6 out of 23 or 26 % habitat characteristics variables being significantly different, leaving out 17 out of 23 or 74 % of habitat characteristic variables, and 5 out of 8 derived habitat components, that do not significantly differ within eucalyptus plantation across the study sites. It is expected in an exotic plantation to have similarity in vegetation characteristics owing to the human intervention rather than natural processes taking place (e.g., Bettinger et al., 2016). It also not uncommon to find homogeneous characteristics in a plantation due to selection of planted trees, i.e., planted at the same time, similar growth characteristics, and similar silvicultural practices imposed (e.g., Evans, 2000). Marian et al. (2020) reported that areas reforested by planting monoculture timber plantations in Andean montane forests resulted in a reduced above ground diversity as compared to in primary forests. Lamb et al. (2005) also points out that although monoculture plantations are successful in providing local communities with timber, they have indeed led to the homogenisation of the landscape. Similar findings were reported by Hobbs et al. (2002) that eucalyptus plantation portrays

relative structural simplicity. It is therefore evident that although eucalyptus plantation is now common within the study area, and potentially spreading to new areas, it is more homogenous and lacks structural heterogeneity relative to natural forests. It probably only serves as an additional forest and habitat type, that just complements the natural forest, but could not replace the characteristics and conditions found in natural forests.

4.4.2 Undisturbed, disturbed and eucalyptus plantation as valid classification of forest types.

Undisturbed forest type in this study is defined by habitat characteristics that are typical of complex forest with less or no disturbance (Table 4.7). This included presence of large tree size, high vegetation cover above 5 m and high canopy cover, deep litter depths, and existence of snags. These habitat characteristics also varied highly within undisturbed forest, showing greater habitat complexity. The presence of particularly high measures of dead wood (snags) and high canopy cover indicates a less disturbed forest (Wirth et al., 2009). These also agree with descriptions given by White and Lloyd (1995) for the definition of old-growth forests as "that old growth encompasses the later stages of forest stand development that typically differ from earlier stages in a variety of characteristics which may include tree size, accumulations of large dead woody material, number of canopy layers, species composition, and ecosystem function". Undisturbed forest type also occurred at a greater distance from the forest boundary (approx. 3767 ± 41.80 m), and at high altitude of 2098 ± 38.00 m a.s.l. This probably also contributed to having less disturbance incidences. These features qualify this forest type to be classified as undisturbed forest relative to the other forest types studied.

Disturbed forest type on the other hand is characterised by mainly significantly high vegetation cover between 1-5 m, canopy height at between 11 and > 20 m, high number of saplings, high number of fallen trees and high FHD. These characteristics portrays a forest with more open canopy that allows more sunlight and likely development of undergrowth (Théry, 2001), probably including development and establishment of saplings. Hitimana et al. (2004) described disturbed forest sites in Mount Elgon Forest in East Africa (also Afromontane forest) as having a forest canopy characterised by

dispersed foliage and more canopy gaps that led to thick and continuous undergrowth. Lack of large trees in the disturbed forest type in this study signify a disturbance through logging. This is because forest structure naturally varies with age (Lin et al., 2018), and disturbance (Graefe et al., 2020). For instance, diameter distributions are commonly used to assess the disturbance effect within forests (e.g., Denslow, 1995) and to detect trends in regeneration patterns (Poorter et al., 1996). Apart from other factors that could be responsible for treefall in tropical forests such as rocky and shallow soils, weakness of trees from disease and insect attack, treefall has been highlighted to be more in exposed sites, where the crown cover of trees was not closed (Arriaga, 2000).

Eucalyptus plantation forest type is characterised by significantly taller trees, dispersed trees, thick near ground vegetation at 0-1 m, and more cut trees and number of trails. Because of silvicultural activities carried out within eucalyptus plantation as a management operation, it is prone to high number of cut trees and trails. However, the significantly high vegetation thickness at 0-1 m could be because of less canopy cover that allows more sunlight to reach the forest floor and encouraging herbaceous and grassy vegetation growth. The silvicultural spacing of planted trees leads to high tree-to-tree distance (dispersed trees), while the nature and growth characteristic of eucalyptus trees has made them exceptionally taller than trees in natural forests. These exceptional characteristics of eucalyptus plantation are new and unique to it in the study area.

In general, each forest type has inherent habitat characteristics, mostly composed of vegetation structures. The characteristics place each of the forests as a unique forest entity, and therefore classification of the studied forest as undisturbed, disturbed and plantation forest types is valid. The role of these forest types and their habitat characteristics in biodiversity conservation is assessed in the subsequent chapters.

4.4.3 Management and conservation implications

Management of both natural forests and exotic plantations in Africa is constrained by limited understanding of inherent structural characteristics and ecological roles of these forests. This is particularly problematic where the forest management has introduced participatory forest management approaches that encourage local community use of forest

resources that are likely to cause forest disturbance. Proper conservation and management have been significantly constrained by absence of scientific information on existing forests and forest type. Management plans for both wildlife and forest conservation in Afromontane forest of Kenya have highlighted the need for updated and current scientific research to guide their management and conservation strategies in the future (KFS, 2010; KWS, 2010). For the sake of conservation and management of these forests, the current chapter has highlighted new information on variation of habitat characteristics within, and between three existing broad forest types. For example, it has determined the validity of the use of the three forest type terms undisturbed, disturbed and eucalyptus plantation, and these terms can be used as defined within the scope of this study. The study has revealed that the difference in forest types is based on forest structural attributes (habitat characteristics). Undisturbed forest type is more variable in terms of habitat structural characteristics and habitat complexity variables while disturbed forest type has more habitat diversity arising from local disturbance which act mostly on foliage-based structures. Eucalyptus plantation are more homogeneous based on its vegetation structural arrangement and how it is planted and subjected silvicultural activities. In addition, the native vegetation growing underneath the eucalyptus plantation forest, serves as additional habitat structures in the plantation.

The management of Afromontane forests like Mount Kenya forest reserve should be aware that the difference in habitat characteristics between undisturbed and disturbed forest types is because of the local anthropogenic disturbance. The disturbance serves to reduce variability in vegetation structures and complexity of forest, but perhaps it can enhance diversity of vegetation arrangement creating more niche openings (i.e., as seen in foliage height diversity). But homogenisation and simplification of habitat conditions are not conducive for many organisms and therefore management efforts need to be made to ensure that anthropogenic activities that affect forest structure within these sites are minimised. Activities such as placement of hives (most of which were seen ready for placement at Chuka forest site (Figure 4.4 a)), livestock feed harvesting (Figure 4.1 b), fuelwood collection, especially felling of snags, all serve to emphasise the local community dependency on Mount Kenya forest reserve disturbed site.
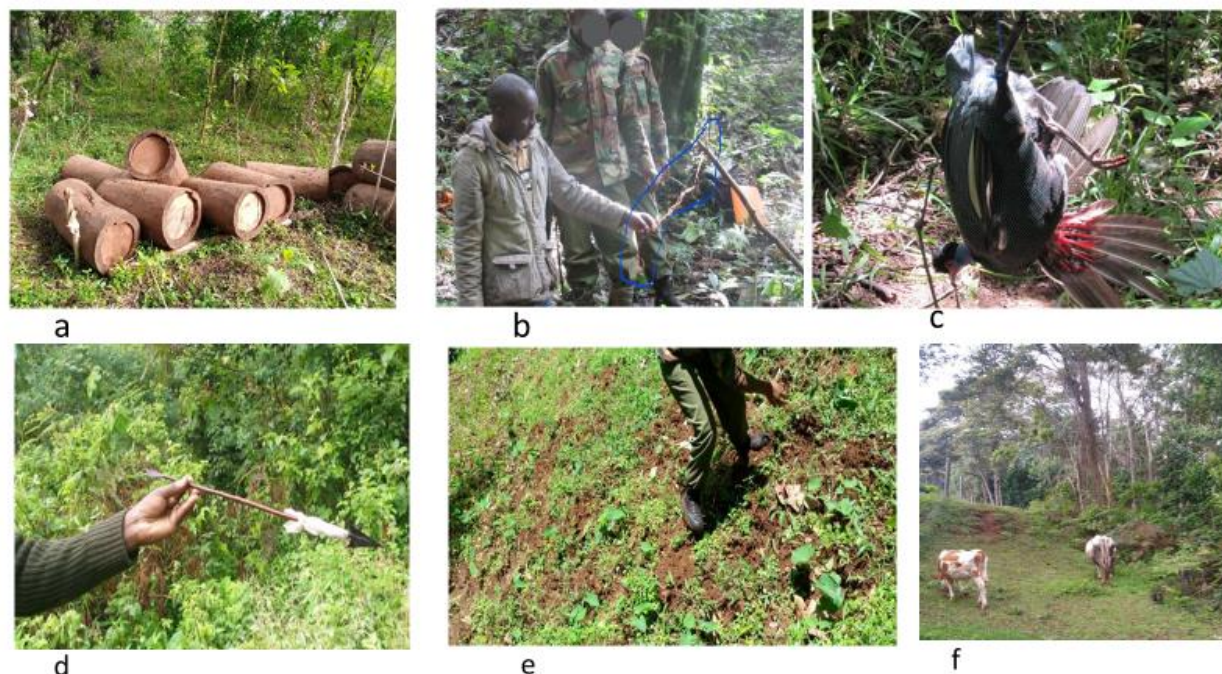
Figure 4.4: (a) Traditional bee hives for local community members ready to be placed at Chuka disturbed forest site. Photo taken 23/02/2019 (b) A snare found by researcher and forest security guards during field data collection at Chuka disturbed forest site. Photo taken on 17/10/2019 (c) A snared Crested guineafowl (*Guttera pucherani)* found and rescued with wing injuries near a point count at Ruthumbi disturbed forest site. Photo taken on 23/02/2020 (d) A recovered (suspected) poisoned arrow at Chuka disturbed forest site. Photo taken on 12/03/2020 (e) A small, cultivated farm with assorted food plants including sweet-potatoes *(Ipomoea batatas*) near a stream at Chuka disturbed forest site (with a forest guard uprooting the plants). Photo taken on 15/10/2019 (f) Livestock grazing near a forest boundary but within the protected forest at Castle disturbed forest site. Photo taken on 4/05/2019.

Only undisturbed forest type was characterised by high number of snags in this study. Snags might have been reduced in the disturbed forest type through removal as sources of fuelwood. This removal of snags can have detrimental effects on other organisms dependent on snags such as illustrated in Chapter 2, Figure 2.2. To manage and conserve these, forest managers, and local communities that are permitted to collect fuelwood from the forest should agree and place appropriate measures that stop or restrict collection of fuelwoods to only fallen branches and not felling dead standing snags nor fallen and rotting logs.

Different habitat structural characteristics across forest types depicts uniqueness of forest types and shows the importance of Afromontane forest ecosystem as a structurally

heterogenous landscape. Conservation of the whole ecosystem ensures enhanced diverse structural landscape protection, potentially crucial for overall biodiversity conservation. The eucalyptus plantation for example, is evidently adding some habitat characteristics to the landscape that may be complementary to the existing natural forest characteristics. However, the contribution of eucalyptus plantation for snags, or for species or organisms that require snags is poor. In addition, eucalyptus could be lacking in other dietary support for other organisms, for instance lack of fruits availability for frugivorous birds make it unsuitable for use by these birds. The studied habitat features in the current study may be continuously affected by local disturbances and may need to be regularly monitored to determine changes over time, and to document the dynamics of habitat structural characteristics based on the management and conservation strategies that may be adopted in future.

4.4.4 Conclusion and recommendations

Most habitat charactersitics vary significantly across all forest types, thus they are uniquely different from each other, making the classification into undisturbed, disturbed and plantation valid. Undisturbed forest type in Mount Kenya Afromontane is characterised by habitat charactersitics depicting forest complexity while disturbed forest is less structurally complex, yet it has diversity of habitat characteristics because of local disturbance that have enhanced foliage strutural diversity instead. That is, apart from reducing forest complexity in disturbed forest type, the local controlled use of Afromontane forests by adjacent communities do not pose serious destruction of habitat vegetation, and it is recommended that it continues to benefit the community.

In this study, Eucalyptus plantation's habitat characteristics is most homogenous (thus simple) and characterised by structural characteristics that are new to the Afromontane ecosystem such as tallest dispersed trees, dense near ground vegetation that consist of native vegetation as undergrowth. Therefore, the use of eucalyptus plantation for purposes of rehabilitation of degraded sites and to increase overall forest cover may continue but care should be taken to ensure it only plays a complementary role because it's simple habitat characteristics cannot replace the natural forest characteristics. The use of exotic

eucalyptus for rehabilitation purposes should cautiously be used and only after it has proved difficult to obtain an alternative fast growing local tree species. This is because, elsewhere, the use of exotic eucalyptus has not been good to the environment especially under drying climates (Gonçalves et al., 2017; Bayle, 2019).   The role of each forest type and its' habitat characteristics on the diversity and community composition of tropical montane birds is assessed in the subsequent chapters.

Chapter 5- The value of undisturbed, disturbed and plantation forests for Mount Kenyan bird diversity

5.1 Introduction

Tropical forests cover 15 % of the Earth's surface and harbour 80 % of known terrestrial species (Rajpar, 2018). The forests are home to most endemic, rare, and threatened species including bird that play a significant role in ecosystem functioning, such as pollination, seed dispersal and control of pests (Rajpar, 2018). But tropical forests are threatened by deforestation driven by global demands for timber, paper and lands for crops and settlements (Lewis et al., 2015; Birdlife International, 2021). In protected tropical forests, although there are general positive effects of protections on the diversity of bird species (e.g., Cazalis et al., 2020), increasing human populations, and human demands on forest resources, and emerging forest management approaches threatens effectiveness of tropical forest in protecting forest habitats (Kareiva et al., 2007; Bradshaw et al., 2009; Nasi and Frost, 2009; Chisika and Yeom, 2021). Effective forest protection maintains characteristics of intact undisturbed forests in terms of habitat characteristics and vegetation structures that are useful for birds (Cazalis et al., 2020), but with human access and use of forests, it creates habitat disturbances (leading to disturbed habitats) (Kleinschroth et al., 2019; Angelstam et al., 2021). On the other hand, the occurrence of diverse land use systems also created by humans (e.g., agroforests, plantations, agricultural landscapes) create other types of ecosystems that are likely to modify protection of bird species in the tropics (Farwig et al., 2008). But conservation values of the resultant forest types (e.g., undisturbed, disturbed and plantations), in terms of diversity of birds as influenced by habitat characteristics and vegetation structures remain a significant knowledge gap. Understanding of the conservation values of these emerging forest types in the tropics is of fundamental importance to enable the managers, policy makers and conservationists to formulate effective conservation approaches to enhance positive contribution of all forest types.

Natural tropical forests in general are of high conservation values for birds (Douglas et al., 2014; Mahiga et al., 2019), but when disturbed its quality declines and conservation

values deteriorate both in habitat quality (i.e., characteristics and complexity) and in forest bird species diversity (Barlow et al., 2016). For instance, undisturbed, disturbed, agroforests, and plantation forest types are characterised by varying habitat structural characteristics (e.g., Chapter 4) (also Hitimana et al., 2004) in the tropics. The forest types and habitat characteristics can mediate modification of bird species diversity and their functional groups (Bett et al., 2016; Jung et al., 2017; Levey et al., 2021). For example, the level of habitat diversity has been found to influence the diversity of bird species in forests (Rompré et al., 2007; Campbell et al., 2021), i.e., highly diverse habitats can hold high species diversity (MacArthur and MacArthur, 1961; Mulwa et al., 2012). Similarly, occurrence of feeding guilds can be influenced by the habitat characteristics that ensures availability, diversity, and possible access to foraging materials (i.e., availability of array of flowering plants, fruiting trees, grain and seed-bearing plants and prey) (Koen et al., 1988; Tchoumbou et al., 2020). Since habitat characteristics are also related to forests' vegetation structural arrangement (how different parts of trees are arranged, especially within different forest types), bird species diversity can be associated with the presence and diversity of vegetation structural arrangements, as predicted by vegetation structure hypothesis (Hurlbert, 2004; Tews et al., 2004). Thus, foliage diversity or canopy cover for example, that differs in forest types may drives bird species diversities (Hurlbert, 2004; Fajardo and Gundale, 2018). But habitat characteristics and vegetation structural arrangement is mainly driven by human actions (i.e., through removals in forests and plantings exotic plantations) and thus may create forest types that influence bird species diversity.

Any form of forest simplification that affects its habitats (either through disturbance or establishment of monoculture plantations) have consequences not only on forest habitat characteristics as seen in Chapter 4, but are also likely to affects bird diversity in the tropics (Betts et al., 2022). Habitat simplification in forests, either via direct removal of vegetation or poor regeneration can reduce availability of a range of resources: food, shelter, cover, and breeding grounds for birds. This is because modification of vegetation structures and reduction in heterogeneity, interferes with resources and microclimates, presenting a threat to general biodiversity (Chen et al., 1999) but particularly to birds because they are the most sensitive taxa to habitat change (Hatfield et al., 2018; Sherry et

al., 2021). For instance, forest specialists, insectivores, frugivores, nectarivores and species with restricted ranges are particularly prone to decline or local extinction following forest disturbance because they are sensitive, more specialised and highly dependent on undisturbed rather than disturbed tropical forests (Gray et al., 2007; Mulwa et al., 2021). Simplification of habitats not only limit the diversity of forest bird assemblages (i.e., taxonomic groups) but also habitat usage by distinct bird functional groups (forest dependency, dietary groups etc) (St. Pierre and Kovalenko, 2014; Batisteli et al., 2018). For example, forests with reduced habitat structural complexity and niches may also experience reduced pollination and seed dispersal capabilities due to few available pollinators and seed dispersers, and consequently poor vegetation regenerations (Breitbach et al., 2012).

Direct effect of human activities resulting from access and use of forest resources degrade and simplify Afrotropical forests (see Asefa et al., 2017; Teucher et al., 2020). Perhaps the effects of human actions such as selective logging and collection of firewood is immediate, for example loss of cover or shelter can immediately expose species to dangers hence drives them to other areas (Tuomainen and Candolin, 2011), thus affecting diversity. It can then lead to consequences that can last longer and probably surpasses the loss from pollination and seed dispersal disruptions by many folds (Malhi et al., 2014; Nuñez et al., 2019). Fuelwood collection for instance, is reported to be one of the most pervasive drivers of forest degradation across Afrotropical forests, and accounts for over 90 % of wood removals in sub-Saharan Africa (Sassen et al., 2015). This, plus other human forest habitat destructive actions like fodder collection, selective logging, tree cuttings in preparation for hive placement in forests (see Figure 2.5 (a) in Chapter 2 and Figure 4.1 in Chapter 4), and establishment of exotic plantations can lead to loss of habitat characteristics and associated qualities such as the sources of foraging materials, cover, and nesting for birds (McKinney and Lockwood, 1999). Establishment of exotic plantation can introduce new vegetation structural characteristics to the environment, but the loss of native forest habitat characteristics and vegetation structures can lead to further loss of forest birds (Goded et al., 2019; Mulwa et al., 2021).

Loss of forest birds through anthropogenic disturbance and emergence of exotic plantations in Afrotropical forests is likely to happen if these disturbances are associated with changes (i.e., loss or decline) in food resources, cover, and shelter availability (Stratford and Şekercioğlu, 2015; Nagy, 2001; Latimer and Zuckerberg, 2021). This is because birds have been found to be sensitive to changes in their habitats (Hatfield et al., 2018; Sherry, 2021; Rurangwa et al., 2021), and different birds may be affected differently, and they can respond to the changes differently. For instance, birds can respond to habitat changes by movement to other suitable areas and/or localised extinction may occur (Temple and Wiens, 1989). However, disturbance, and creation of plantations may also create opportunities for some species or their functional groups to come in from other ecosystems, while making others to thrive rather than be displaced. In cases where there is loss of forest birds (including their functional groups) following disturbance and emergence of exotic plantations, it is consequently followed by loss of critical ecological services (e.g., Whelan et al., 2008) provided by different groups of birds as shown in Table 5.1, alongside other critical benefits as also shown in Table 5.1.

Table 5.1: Ecological contributions and consequences of loss of forest birds and dietary functional groups (Source: adapted from Şekercioğlu, et al., 2004).

Functional group	Ecological process	Ecosystem service and economic benefits	Negative consequences of loss of functional group
All species	Miscellaneous	Environmental monitoring; indirect effects; birdwatching tourism; reduction of agricultural residue and pests; cultural and economic uses	Losses of socioeconomic resources and environmental monitors; unpredictable consequences
Frugivores	Seed dispersal	Removal of seeds from parent tree; escape from seed predators; improved germination; increased economical yield; increased gene flow; recolonization and restoration of disturbed ecosystems	Disruption of dispersal mutualisms; reduced seed removal; clumping of seeds under parent tree; increased seed predation; reduced recruitment; reduced gene flow and germination; reduction or extinction of dependent species
Nectarivores	Pollination	Outbreeding of dependent and or economically important species	Pollinator limitation; inbreeding and reduced fruit yield; evolutionary consequences; extinction
Scavengers	Consumption of carrion	Removal of carcasses; leading other scavengers to carcasses; nutrient recycling; sanitation	Slower decomposition; increases in carcasses; increases in undesirable species; disease

Functional group	Ecological process	Ecosystem service and economic benefits	Negative consequences of loss of functional group
			outbreaks; changes in cultural practices
Insectivores	Predation on invertebrates	Control of insect populations; reduced plant damage; alternative to pesticides	Loss of natural pest control; pest outbreaks; crop losses; trophic cascades
Raptors	Predation on vertebrates	Regulation of rodent populations; secondary dispersal	Rodent pest outbreaks; trophic cascades; indirect effects

In Afrotropical forests, human activities (mainly acting on habitats) are responsible for major bird declines in Afrotropical forests (Borghesio, 2008; Bett et al., 2016; Bett et al., 2017). High demand for forest resources by humans causes great pressure on forests and habitats (Dirzo and Raven, 2003), leading to habitat modification (Foley et al., 2005; Okumu and Muchapondwa, 2020). Lowland Afromontane part of Afrotropical forest habitats are particularly modified, degraded, or lost (Pfeifer et al., 2012 (a); Bobadoye et al., 2017). But this modification arises partly through illegal (Leaver et al., 2019; Kayombo et al., 2020) and permitted access to protected forests (Mkhai et al., 2017; Contreras-Hermosilla, 2000). Permitted access and use (extractive forest products and services) is a phenomenon that have resulted from a shift in forest conservation approach. It occurs where local communities, through participatory forest management (PFM) under the umbrella of Community Forest Associations (CFAs), are allowed to obtain controlled resources such as firewood, grazing materials, medicinal plants among others (see Figure 2.5, Figure 4.1 b and Appendix 1) (Schreckenberg and Luttrell, 2009; Sonkoyo, 2014). The manner of community use of forest and how the members help in forest conservation is formally agreed by members of CFAs and meant to control the excessive use or to check illegal practices among members (Wily, 2002; Tsegaye et al., 2007; Gobeze et al., 2009; Magessa et al., 2020; Okumu and Muchapondwa, 2020). PFM approach was adopted from the early 1990s as a way of addressing challenges of forest degradation and conservation of biodiversity without compromising the objective of rural poverty alleviation (Lund and Nielsen, 2005). PFM has since gained popularity within Afrotropical forests and governments, and forest management agencies have widely adopted it (German, 2009). However, despite many studies in Afrotropical montane (e.g., Tattersfield et al., 2001; Ndegwa, 2014; Kioko et al., 2016; Asefa et al., 2017; Mahiga et al., 2019; Jemal et al.,

2020; Dinesen et al., 2022), how extraction of forest resources with associated change in forest habitat characteristics and vegetation structure affects bird species diversity has not been fully investigated. In addition, conservation role of exotic plantations which is management practices under PFM involving restoration of degraded forest areas (Wily, 2002) is still under contestation in tropical areas (e.g., Hughes, 1994; Chamshama and Nwonwu, 2004; Zerga et al., 2021).

The aim of this study is to examine the impacts of human disturbance and exotic plantation on avian diversity in the lower Afromontane forests of Mount Kenya. Specifically, the impact of these is assessed by comparing avian species richness and abundance across undisturbed, disturbed and eucalyptus plantation forest types. This is done for (1) the entire assemblage, (2) species grouped by forest dependency and dietary guild, and (3) species restricted to Afrotropical highlands biome (ATHB) (i.e., for the species representing species of conservation concern). In addition, this study investigates prediction of habitat characteristics on avian diversity in lower afromontane forest of Mount Kenya.

It is predicted that:
(i) Undisturbed forest and its habitat characteristics, compared with either disturbed or plantation forest types, would contain and predicts higher overall bird species richness and abundance, as well as higher richness and abundance of forest specialists, frugivores, insectivores, nectarivores and species restricted to Afrotropical highlands biome. This is because undisturbed forest has greatest habitat heterogeneity and niches that attracts sensitive and specialised birds.
(ii) Disturbed forest and its habitat characteristics, compared with either undisturbed or plantation forest types would predict lower overall species diversity but higher species richness and abundance of forest generalists, forest visitors, carnivores, and omnivores. This is because disturbed forest has relatively less heterogeneity/niches, has more open canopy than undisturbed forest, and disturbed conditions and edge effects are expected to attract forest generalists, forest visitors, and omnivores. It is expected to attract carnivore bird species which require open space to scan for prey.

(iii) plantation forest, compared with either undisturbed or disturbed forest types, is predicted to have lowest species diversity, higher species richness and abundance of non-forest birds, carnivores, granivores and omnivores birds owing to reduced forest cover and thus more open spaces for predator species. open canopies encourage more light penetration for grain-bearing grassy undergrowth development attract granivore birds, and more non-forest conditions attracting non-forest birds.

5.2 Methods

5.2.1 Study area and selection of study sites

The study was conducted within the eastern, southeastern, and eastern foothills of Mount Kenya. Specifically, three study sites were selected each containing disturbed, undisturbed and eucalyptus plantation forests. Further detail about the study area, study sites and forest types can be found in Chapter 3 (sections 3.1 and 3.2).

5.2.2 Bird sampling and classification

Point counts bird sampling (laid out as described in Chapter 3 (section 3.3)) were conducted across four seasons (long rainy season (April-June), short dry season (July-Sept), short rainy season (October-December), and long dry season (January-March) between April 2019 to March 2020 in order to obtain species richness and abundance data. The preparation of bird data prior to analysis for this chapter is detailed in Chapter 3 section 3.3.1 a.

There are 70 species that have been recorded in Kenya that belong to Afrotropical highlands biome (ATHB) species, with only 54 species having been recorded in Mount Kenya (Evans and Fishpool, 2001). For the birds classified as Kenyan Mountains Endemic Birds (KMEB) under species of conservation concern, only eight species have been recorded in Mount Kenya (Evans and Fishpool, 2001). These include: *Francolinus jacksoni* (Jackson's francolin), *Macronyx sharpei* (Sharpe's longclaw), *Turdoides hindei* (Hinde's babbler), *Cisticola hunteri* (Hunter's Cisticola), *Cisticola aberdare* (Aberdare cisticola), *Euplectes jacksoni* (Jackson's widowbird), *Poeoptera kenricki* (Kenrick's starling), and *Cinnyricinclus femoralis* (Abbott's starling).

5.2.3 Habitat characteristics

Habitat characteristics, broken down into measures of vegetation structure, human disturbance, and altitude, slope and distance from forest edge (Table 5.2) were collected within a radius of 10 m (≈0.04 ha) surrounding each point count station where birds' data were collected. Details for sampling protocol, collection, definition, and justifications for each of the variables is described in Chapter 3 section 3.3.1 (b) and Appendix 2.

Table 5.2: Habitat characteristics and how it has been broken down to vegetation structure, human disturbance and altitude, slope, and distance from forest edge. See Appendix 2 for full definitions of each variable.

Habitat characteristics		
Vegetation structure	*Human disturbance*	
Tree height	Number of cut trees	Slope
Tree diameter at breast height (DBH)	Number of trails	Altitude
Tree distance	Number of fallen trees	Distance from forest edge
Low canopy height at 0-10 m		
Mid canopy height at 11-20 m		
High canopy height at > 20 m		
Litter depth		
Foliage height diversity		
Percentage vegetation cover at 0-1 m		
Percentage vegetation cover at 1-3 m		
Percentage vegetation cover at 3-5 m		
Percentage vegetation cover at 5-8 m		
Percentage vegetation cover at > 8 m		
Percentage canopy cover		
Number of saplings		
Number of snags		

5.2.4 Statistical analysis

(i) Species richness and abundance across forest types

Observed bird species richness (S(obs)), estimated species richness (S(est)), and bird abundance were quantified for overall bird assemblage for each forest - (i) undisturbed forest (N = 75 point counts), (ii) disturbed forest (N = 75 points counts), and (iii) eucalyptus plantation forest (N = 40 point counts). Observed and estimated species richness and bird abundance were also determined for each forest dependency groups, dietary guild, and for species restricted to Afrotropical highlands biome (ATHB).

Following Asefa et al (2017), individual-based rarefaction and extrapolation was used to derive S(est) for undisturbed, disturbed and plantation forest types using EstimateS 9.1.0 software (http://viceroy.eeb.uconn.edu/estimates; Colwell, 2012). The reason for estimating species richness was to account for species present but not detected due to rarity (Colwell et al., 2012). Individual-based rarefaction and extrapolation was considered appropriate instead of a sample-based approach because the primary interest here was to estimate and compare species richness across forest types rather than species density; the latter of which is computed from sample-based data (Colwell et al., 2012; Asefa et al., 2017). The use of rarefaction and extrapolation also allows species richness to be compared across forest types at equivalent abundance, because rarefaction standardises the sample size and can be used to calculate species richness before comparing it using a grouping variable or before correlating to an environmental factor (Colwell et al., 2012). The Chao 1 estimator (an appropriate estimator for individual-based data; Colwell et al., 2012) was used to estimate asymptotic species richness (S(est), the total number of species expected in an area, including those species not observed during the survey period) for each forest type to assess sampling completeness. The summed abundance for each forest type of the number of individuals of each species recorded in each point count was used as the input for the individual-based richness computation following Asefa et al. (2017).

Species accumulation (rarefaction) curves for each forest type were computed using the online R-based version of iNEXT (https://chao.shinyapps.io/iNEXTOnline/, Chao and

Colwell, 2017). The curves were computed with 95 % confidence intervals (CI) to enable a robust direct statistical comparison among forest types of extrapolated samples (Colwell et al., 2012, Asefa et al., 2017). Following Asefa et al. (2017), for comparison of forest types of estimated species richness based on extrapolations, smaller samples were extrapolated to the total number of individuals recorded in the larger sample (Colwell et al., 2012).

Observed species richness and total abundance of bird species belonging to each forest dependency category, dietary guild, and the ATHB conservation concern group were statistically compared (using sample coverage) and visually compared (using species accumulation curves, assessing if satisfactorily reaching or approaching asymptote) across forest types. Kenyan mountain endemic birds (KMEB), IUCN Red List threatened species and omnivores were not compared this way owing to their small sample size. Sample coverage, which is a measure of sample completeness, was used to determine how complete the sample was using the online R-based version of iNEXT. Sample coverage gives the proportion of the total number of individuals in a community that belongs to the species represented in the sample (Chao and Jost, 2012). Subtracting the sample coverage from one, gives the proportion of the community belonging to unsampled species (the probability that a new, previously unsampled species would be found if the sample were enlarged by one individual (Chao and Jost, 2012)).

The effect of forest type on species richness and abundance were then tested using generalised linear models (GLMs), all with a Poisson probability distribution and log-link function and performed within SPSS 26 (IBM, Armonk, New York, USA; Guisan et al., 2002). Bird species richness and abundance per point count station were considered response variables, respectively, while forest type was treated as a fixed effect. After GLM analyses, post-hoc pair-wise comparisons across forest type were conducted using Least Significant Difference (LSD). The Wald test was used in this comparison of effect of forest types on species richness and abundance because it is a more broadly applicable test than the Likelihood Ratio Test and can be run with a single model (Glen, 2016). It is also more applicable where there are large sample sizes (Agresti, 1990). Model fit was examined for all models produced, i.e., overall species assemblage, and for forest

dependency, dietary guild and ATHB groups with plots of fitted data vs. standardised model residuals. Results are presented as mean +/- SE, and the statistical significance of differences was tested at $\alpha= 0.05$.

(ii) Prediction of bird species richness and abundance by habitat characteristics across forest types.

As detailed in Chapter 4 section 4.3.3 (a) among predictor variables, tree basal area was dropped from further analysis (Table 5.2) since it was collinear with tree DBH. Tree DBH was retained because basal area is derivative of it.   Using the remaining habitat characteristics as predictor variables (see table 5.2), Poisson and negative binomial regressions were performed to determine the prediction of habitat characteristics on bird species richness and abundance. But before this, a likelihood ratio test of whether the predictor variables collectively improve the model over the intercept-only model (i.e., with no predictor variables added) were assessed (Appendix 6 a, b and c). In addition, in order to make valid inferences from the Poisson regression, the residuals of the regression were checked for normality (Appendix 7).

Poisson and negative binomial regressions with log link were then used to assess specific predictor variables that predicted species richness and abundance response variables. This is because both response variables are count data, but Poisson or negative binomial regressions were selected depending on the nature of the data and the underlying statistical assumption of equal means and variance (i.e., if equi-dispersed, over-dispersed or under-dispersed). To test the required assumptions of equi-dispersion for each Poisson regression, the goodness of fit table was assessed by looking at the ratio of value of deviance and degree of freedom (Cameron and Trivedi, 2013). If the value is greater than one it shows that data are over-dispersed, if less than one is under-dispersed and if equal to one is equi-dispersed.

The problem of overdispersion in the response variables were addressed by using negative binomial logistic regression with log link function instead of Poisson regression (Payne et al., 2018) for the analysis. The problem of under-dispersion was not an issue since the ratio of value of deviance and degree of freedom in the goodness of fit table was closer to

one (i.e., ≥ 0.88), and Poisson regression analysis was carried out instead. Table 5.3 shows the dependent variables with levels of dispersion i.e., equi-dispersion, over-dispersed, or under-dispersed and the type of regression model used. In the case of predicting species richness and abundance by habitat characteristics, the Likelihood ratio test was used in Poisson and negative binomial since it is a more powerful test of regression parameters than Wald's test (Cameron and Trivedi, 2013).

Table 5.3: The response variables which are count data showing the one which are either equidispersed, under-dispersed, or over-dispersed (as indicated by a tick) and the kind of regression model used to analyse it. FS = forest specialists, FG = forest generalists, FV = forest visitors, NF = non-forest birds, ATHB = Afrotropical highlands biome restricted species.

Dependent variable	Equidispersed	under-dispersed	over-dispersed	Regression model
Overall richness		✓		Poisson
Overall abundance			✓	Negative binomial
FS richness			✓	Negative binomial
FS abundance			✓	Negative binomial
FG richness		✓		Poisson
FG abundance			✓	Negative binomial
FV richness			✓	Negative binomial
FV abundance			✓	Negative binomial
NF richness			✓	Negative binomial
NF abundance			✓	Negative binomial
Carnivore richness	✓			Poisson
Carnivore abundance	✓			Poisson
Frugivore richness	✓			Poisson
Frugivore abundance			✓	Negative binomial
Granivore richness	✓			Poisson
Granivore abundance			✓	Negative binomial
Insectivore richness		✓		Poisson
Insectivore abundance			✓	Negative binomial
Nectarivore richness	✓			Poisson
Nectarivore abundance	✓			Poisson
Omnivore richness	✓			Poisson
Omnivore abundance			✓	Negative binomial
ATHB richness		✓		Poisson
ATHB abundance			✓	Negative binomial

5.3 Results

5.3.1 Bird species richness and abundance across forest types

(a) All species

A total of 6645 individuals comprising 147 bird species across 49 families were recorded across all forest types throughout the study ($n = 190$). This includes 2523 individuals from 82 species in undisturbed forest ($n = 75$), 2728 individuals from 100 species in disturbed forest ($n = 75$) and 1394 individuals from 94 species in plantation forests ($n = 40$) (Table 5.4; Appendix 8).

Table 5.4: Observed (S(obs)) and estimated [(S(est)) based on Chao 1 and S(est) extrapolated)] species richness, total abundance, mean species richness and abundance per point count within undisturbed, disturbed and eucalyptus plantation forest types in Mount Kenya forest reserve. The values for species richness and abundance per point count are mean ± se. The values for estimated richness (S(est)) are mean ± 95 % CI of randomisations (100 randomisations) in each case. Superscripts next to values of S(obs) and abundance with no common letters denote significant difference of S(obs) and abundance across forest types, based on GLMs ($p < 0.05$). Means of species richness and abundance per point count with no common superscript letters in common denote a post-hoc LSD, which is significant between paired forest types ($p < 0.05$). Sample coverage = measure of sample completeness (maximum possible value of 1). % Change in richness = % change between S(obs) and S(est) Chao 1 richness (i.e., (increase/S_{obs}) *100). N is the number of point count stations.

Frst type	N	S (obs)	Abun	Richness per point count	Abundance per point count	S(est) chao 1	S(est) extrapolated	Sample coverage	% Change (richness)
Und	75	82 [a]	2523 [a]	19.09 ± 0.51^{ac}	33.64 ± 0.67^{ac}	82.71 ± 0.18	82.09 ± 0.01	0.99	0.86
Dist	75	100 [b]	2728 [b]	20.29 ± 0.52^{a}	36.37 ± 0.70^{b}	104.17 ± 0.80	100.10 ± 0.01	0.99	4.17
Plan	40	94 [c]	1394 [c]	18.13 ± 0.67^{bc}	34.85 ± 0.93^{bc}	99.50 ± 0.38	97.55 ± 0.04	0.99	5.85

Frst = Forest, Undist = undisturbed, Distur. = disturbed, Planta. = plantation, Abun. = abundance.

The comparison of rarefaction and extrapolated species curves of birds across forest types show that the individual-based curves in all cases approached an asymptote (Figure 5.1). This implies that there were enough samples in all forest types to reliably assess the differences in bird species richness and abundances between them. In addition, assessment of the sample coverage showed 0.99 sample completeness at each forest type

(see Table 5.4). The samples in the three forest types therefore have identical sample coverage of 99 %, implying that they are equally complete, and that the raw data can be directly compared (Chao and Jost, 2012). Moreover, the percentage difference between estimated and observed species richness was less than 6 % in all cases (Table 5.4) showing that only a small/insignificant proportion of species were undetected by the sample.
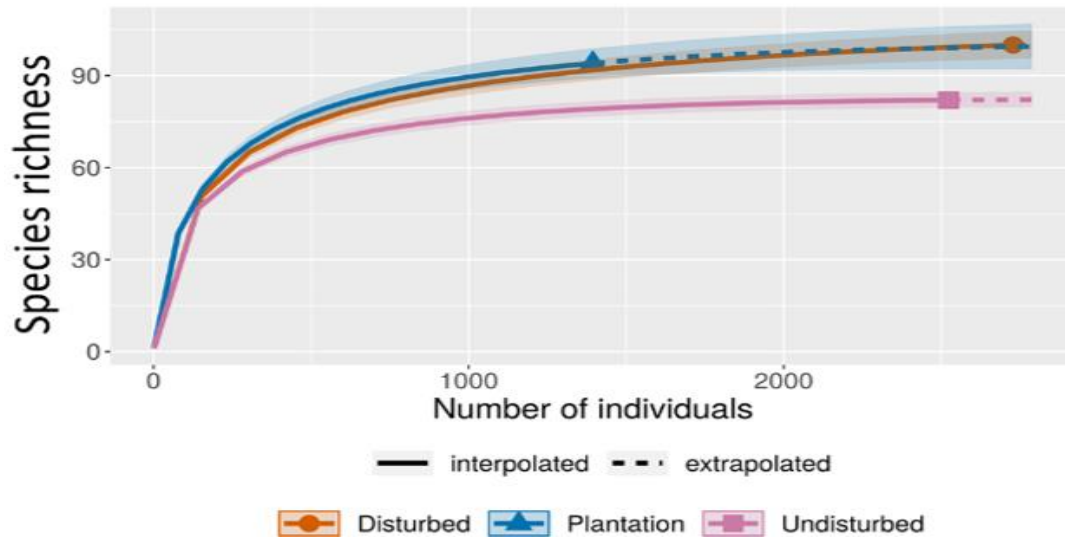
Figure 5.1: Species richness individual-based rarefaction and extrapolation curves for birds in undisturbed, disturbed and plantation forest types, showing approach to an asymptote.

Forest type had a statistically significant effect on overall species richness and abundance (Wald χ^2 = 6.795, df = 2, p = 0.033; Wald χ^2 = 8.030, df = 2, p = 0.018, respectively). Post-hoc pairwise mean differences between the forest types showed species richness to be significantly higher in disturbed forest when compared to plantations (mean difference ± se = 2.17 ± 0.851, df = 1, p = 0.011) and abundance to be significantly higher in disturbed rather than undisturbed forests (mean difference ± se = 2.73 ± 0.966, df = 1, p = 0.005). Mean species richness and abundance per point count were also significantly highest in disturbed forest, with the significantly lowest species richness per point count was found in eucalyptus plantations and significantly lowest abundance per point count in undisturbed forest (Table 5.4).

(b) Forest dependency

Rarefaction and extrapolated species richness curves of the four forest dependency groups within each forest type showed a satisfactory asymptote (Figures 5.2) and sample coverage of $\geq$ 90 % (Table 5.5).
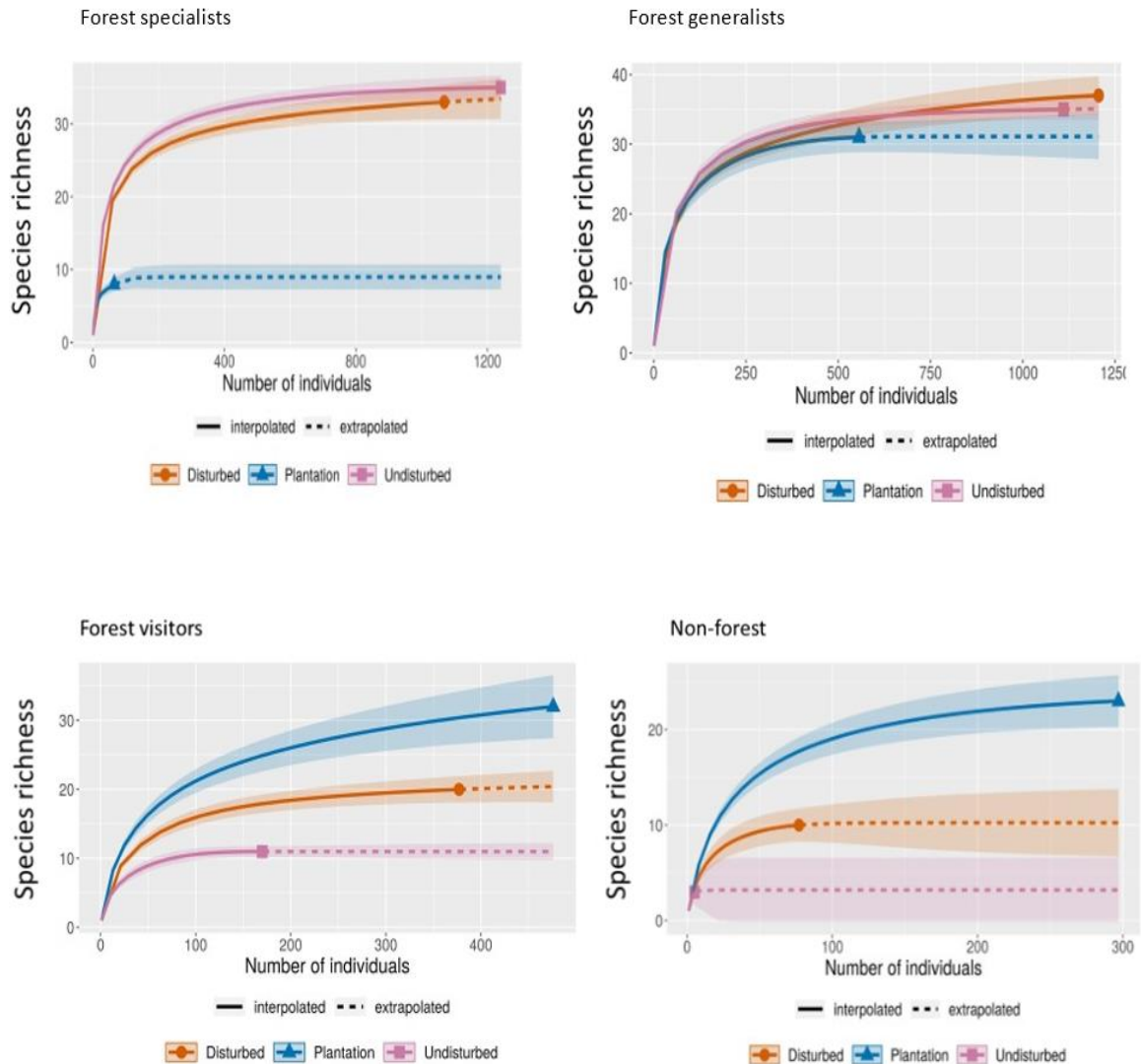
Figure 5.2: Species richness individual-based rarefaction and extrapolation curves for each forest dependency group (forest specialists, forest generalists, forest visitors and non-forest), in undisturbed, disturbed and eucalyptus plantation forests of Mount Kenya forest reserve.

Table 5.5: Observed (S(obs)) and estimated [(S(est)) based on Chao 1 and S(est) extrapolated)] species richness, total abundance, mean species richness and abundance per point count within undisturbed, disturbed and eucalyptus plantation forest for four forest dependency groups (FDG) in Mount Kenya forest reserve. The values for species richness and abundance per point count are mean ± se. The values for estimated richness (S(est)) are mean ± 95 % CI of randomizations (100 randomizations) in each case. Superscripts next to values of S(obs) and abundance with no common letters denote significant difference of S(obs) and abundance of forest dependency groups across forest types, based on GLMs (p < 0.05). Means of species richness and abundance per point count with no superscript letters in common denote a post-hoc Least Significant Difference (LSD), which is significant between paired forest types (p < 0.05). Sample coverage = measure of sample completeness (maximum possible value of 1). % Change in richness = % change between S(obs) and S(est) Chao 1 richness (i.e., (increase/S_{obs}) *100). N is number of point count stations. FD = forest dependency, FS = forest specialists, FG = forest generalists, FV = forest visitors, and NF = non-forest.

Forest type	FD	N	S (obs)	Abun.	Richness per point count	Abundance per point count	S(est) chao 1	S(est) extrapolated	Sample covera.	% Change (richn.)
Undist.	FS	75	35 [a]	1241 [a]	9.95 ± 0.36 [a]	16.55 ± 0.47 [a]	35.00 ± 0.67	35.00 ± 1.94	0.99	0.00
Distur.	FS	75	33 [b]	1069 [b]	8.27 ± 0.33 [b]	14.25 ± 0.44 [b]	34.00 ± 1.82	33.18 ± 5.74	0.99	3.03
Planta.	FS	40	8 [c]	65 [c]	0.98 ± 0.16 [c]	1.63 ± 0.20 [c]	8.98 ± 2.22	8.91 ± 6.22	0.96	12.25
Undist.	FG	75	35 [a]	1110 [a]	7.76 ± 0.32 [a]	14.80 ± 0.44 [a]	35.00 ± 1.00	35.05 ± 2.76	0.99	0.00
Distu.	FG	75	37 [a]	1205 [b]	7.76 ± 0.32 [a]	16.07 ± 0.46 [b]	37.43 ± 5.54	37.00 ± 3.36	0.99	0.00
Planta.	FG	40	31 [a]	556 [c]	7.40 ± 0.43 [a]	13.90 ± 0.59 [a]	31.00 ± 0.34	31.00 ± 1.24	0.99	0.00
Undist.	FV	75	11 [a]	170 [a]	1.37 ± 0.14 [a]	2.27 ± 0.17 [a]	11.00 ± 1.39	11.00 ± 0.00	1.00	0.00
Distu.	FV	75	20 [b]	377 [b]	3.53 ± 0.22 [b]	5.03 ± 0.26 [b]	20.33 ± 5.93	20.99 ± 5.77	0.99	1.65
Planta.	FV	40	32 [c]	476 [c]	6.32 ± 0.40 [c]	11.90 ± 0.55 [c]	37.24 ± 5.37	32.57 ± 10.21	0.98	16.37
Undist.	NF	75	3 [a]	5 [a]	0.01 ± 0.01 [a]	0.03 ± 0.02 [a]	3.00 ± 0.66	3.20 ± 2.40	0.90	0.00
Distu.	NF	75	10 [b]	77 [b]	0.73 ± 0.10 [b]	1.03 ± 0.12 [b]	10.00 ± 0.70	10.25 ± 2.80	0.98	0.00
Planta.	NF	40	23 [c]	297 [c]	3.43 ± 0.30 [c]	7.42 ± 0.43 [c]	23.25 ± 4.77	23.43 ± 3.82	0.99	1.09

Undist. = undisturbed, Distur. = disturbed, Planta. = plantation, Abun. = abundance, richn. = richness.

S(est) species richness across each forest type for each forest dependency group shows either no increase or a slight increase compared to S(obs), and a sample coverage of ≥ 90 % (Table 5.5). The percentage difference between S(est) and S(obs) is ≤ 3.03 in undisturbed and disturbed forest types, for all groups of forest dependency, while the highest increase was recorded within the plantations for forest visitors (16.37 %) and forest specialists (12.25 %) (Table 5.5).

Forest types had a statistically significant effect on the following: (1) forest specialists' species richness and abundance (Wald χ^2 = 201.60, df = 2, p < 0.001; Wald χ^2 = 333.31, df = 2, p < 0.001, respectively), which was significantly highest in undisturbed forest type and lowest in eucalyptus plantation; (2) forest generalists' species abundance (Wald χ^2 = 333.31, df = 2, p < 0.001), which was significantly highest in disturbed forest and lowest in eucalyptus plantation; (3) forest visitors' species richness and abundance (Wald χ^2 = 173.74, df = 2, p < 0.001; Wald χ^2 = 387.52, df = 2, p < 0.001, respectively) which were significantly highest in eucalyptus plantation respectively and lowest in undisturbed forest, and (4) non-forest species richness and abundance (Wald χ^2 = 119.17, df = 2, p < 0.001; Wald χ^2 = 293.61, df = 2, p < 0.001, respectively) which were also significantly highest in eucalyptus plantation respectively and lowest in undisturbed forest (Table 5.5). Similar trend is observed on species richness and abundance per point count for all forest dependency groups (Table 5.5).

(c) Dietary guilds

The species accumulation and extrapolation curves for each dietary guild approached an asymptote for all forest types (Figure 5.3), with satisfactory sample coverage of ≥ 98 %, except for carnivores which had a sample coverage of 60 % in plantation forest (Table 5.6). Carnivore birds were therefore not statistically compared across forest types due to its low sample coverage (Table 5.6). S(est) species richness (based on Chao 1) for carnivorous bird in plantations was more than twice (i.e., 108 %) that of the observed species richness, while for the other dietary guilds, estimated richness was ≤ 4 % different from S(obs) richness (Table 5.6). Species richness and abundance for frugivores, granivores, insectivores and nectarivores across forest types were compared since each had a sample coverage showing sample completeness of ≥ 98 % across forest types.
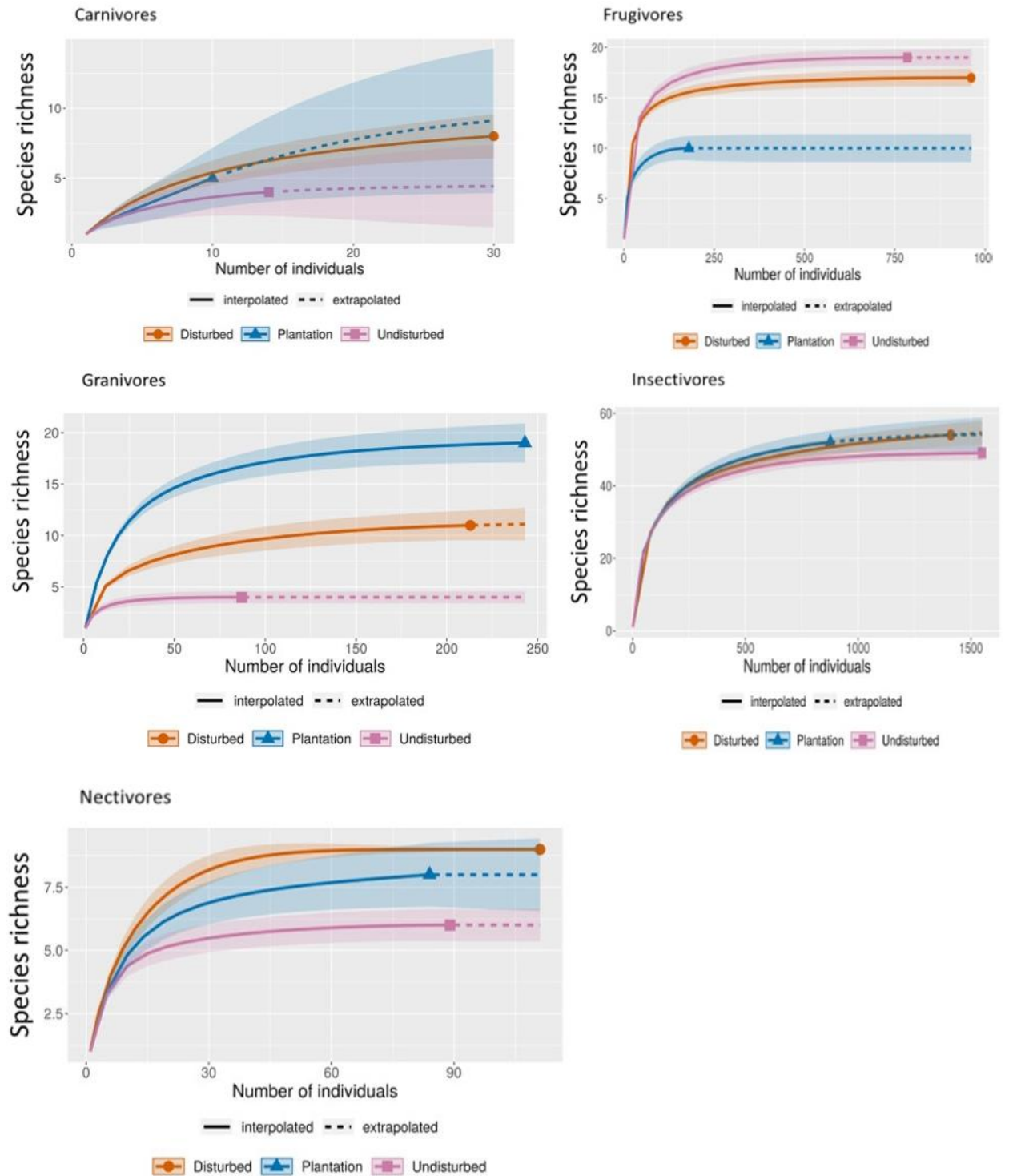
Figure 5.3: Species richness individual-based rarefaction and extrapolation curves for each dietary guild (carnivores, frugivores, granivores, insectivores and nectarivores), in undisturbed, disturbed and plantation forest types.

Table 5.6: Observed (S(obs)) and estimated [(S(est)) based on Chao 1 and S(est) extrapolated)] species richness, total abundance, mean species richness and abundance per point count within undisturbed, disturbed and plantation forest types for dietary guild groups in Mount Kenya forest reserve. The values for species richness and abundance per point count are mean ± se. The values for estimated richness (S(est)) are mean ± 95 % CI of randomizations (100 randomizations) in each case. Superscripts next to values of S(obs) and abundance with no common letters denote significant difference of S(obs) and abundance of dietary guild groups across forest types, based on GLMs (p < 0.05). Means of species richness and abundance per point count with no superscript letters in common denote a post-hoc Least Significant Difference (LSD), which is significant between paired forest types (p < 0.05). Sample coverage = measure of sample completeness (maximum possible value of 1). % Change in richness = % change between S(obs) and S(est) Chao 1 richness (i.e., (increase/S_{obs}) *100). N is number of point count stations.

Forest types	DG	N	S (obs)	Abun.	Richness per point count ± se	Abundance per point count ± se	S(est) chao 1	S(est) extrapo.	Samp -le cove.	% Change (rich.)
Undist.	Carn.	75	4	14	0.17 ± 0.05	0.19 ± 0.05	4.00 ± 1.06	4.30 ± 3.34	0.94	0.00
Distur.	Carn.	75	8	30	0.33 ± 0.07	0.40 ± 0.07	8.32 ± 5.79	8.00 ± 0.00	0.94	4.00
Planta.	Carn,	40	5	10	5.00 ± 1.29	9.50 ± 0.73	10.40 ± 6.33	9.11 ± 3.95	0.60	108.00
Undist.	Frug.	75	19 [a]	784 [a]	5.73 ± 0.28 [a]	10.45 ± 0.37 [a]	19.00 ± 1.05	19.00 ± 0.00	1.00	0.00
Distur.	Frug.	75	17 [b]	961 [b]	6.92 ± 0.30 [b]	12.81 ± 0.41 [b]	17.00 ± 0.88	17.00 ± 0.00	1.00	0.00
Planta.	Frug.	40	10 [c]	180 [c]	2.50 ± 0.25 [c]	4.50 ± 0.34 [c]	10.00 ± 1.33	10.00 ± 0.00	1.00	0.00
Undist.	Gran.	75	4 [a]	87 [a]	0.75 ± 0.10 [a]	1.16 ± 0.12 [a]	4.00 ± 0.48	4.00 ± 0.00	1.00	0.00
Distur.	Gran.	75	11 [b]	213 [b]	1.45 ± 0.14 [b]	2.84 ± 0.20 [b]	11.00 ± 0.69	11.11 ± 1.97	0.99	1.00
Planta.	Gran.	40	19 [c]	243 [c]	2.15 ± 0.23 [c]	6.08 ± 0.39 [c]	19.00 ± 0.67	19.00 ± 1.94	0.99	0.00
Undist.	Insec.	75	48 [a]	1516 [a]	11.12 ± 0.39 [a]	20.21 ± 0.52 [a]	48.00 ± 0.67	48.00 ± 1.96	0.99	0.00
Distur.	Insec.	75	52 [a]	1388 [a]	10.09 ± 0.37 [a]	18.51 ± 0.50 [b]	53.43 ± 1.93	52.21 ± 5.59	0.99	2.75
Planta.	Insec.	40	49 [a]	767 [a]	10.35 ± 0.51 [a]	19.18 ± 0.69 [ab]	50.87 ± 2.25	50.56 ± 7.61	0.99	3.82
Undist.	Necta.	75	6 [a]	89 [a]	0.93 ± 0.11 [a]	1.19 ± 0.13 [a]	6.00 ± 0.83	6.00 ± 0.00	1.00	0.00
Distur.	Necta.	75	9 [a]	111 [b]	1.24 ± 0.13 [ab]	1.48 ± 0.14 [a]	9.00 ± 0.26	9.00 ± 0.00	1.00	0.00
Planta.	Necta.	40	8 [a]	84 [c]	1.40 ± 0.19 [b]	2.10 ± 0.23 [b]	8.00 ± 1.59	8.00 ± 0.00	0.98	0.00

Undist. = undisturbed, Distur. = disturbed, Planta. = plantation, Abun. = abundance, rich. = richness, cove. = coverage, extrapo. = extrapolated, Carn. = carnivores, Frug. = frugivores, Gran. = granivores, Insec. = insectivores, Necta. = nectarivores.

Forest type had a statistically significant effect on species richness and abundance for frugivores (species richness: Wald χ^2 = 87.05, df = 2, p < 0.001; abundance: Wald χ^2 = 166.69, df = 2, p < 0.001 with significantly highest frugivore species richness in undisturbed forest and highest species abundance in disturbed forest type, and significantly lowest frugivore richness and abundance in eucalyptus plantation respectively (Table 5.6). Granivores species richness and abundance was also significantly different across forest types (species richness: Wald χ^2 = 37.98, df = 2, p <

0.001; abundance: Wald $\chi^2 = 189.21$, df =2, p < 0.001), with highest species richness and abundance in eucalyptus plantation and lowest species richness and abundance in undisturbed forest types respectively (Table 5.6). In addition, nectarivores species abundance was significantly different across forest type (Wald $\chi^2 = 14.29$, df = 2, p < 0.001), with highest abundance in disturbed forest type (Table 5.6), and lowest in eucalyptus plantation.

Species richness and abundance per point counts was significantly highest in disturbed forest and lowest in eucalyptus plantation respectively for frugivore bird species. Granivores species richness and abundance were significantly highest in eucalyptus plantation and lowest in undisturbed forest respectively (Table 5.6). Nectarivore species richness and abundance per point count was significantly highest in eucalyptus plantation.

(d) Species of conservation concern

(i) IUCN Red list categories

Only one threatened species was recorded in this study, namely Abbott's starling (*Cinnyricinclus femorali*) (Vulnerable). The remaining species were non-threatened, i.e., either Near Threatened or Least Concern, with the majority (98 %) falling under the latter category (Appendix 8).

(ii) Kenyan Mountain Endemic Birds

Three out of the eight recognised KMEB species (see section 5.2.2) were observed in this study, namely *Cisticola hunteri* (forest generalist), *Poeoptera kenricki* (forest specialist) and *Cinnyricinclus femoralis* (forest specialist). *Cisticola hunteri* (was recorded only within undisturbed forest and plantations, while *Poeoptera kenricki* was recorded within undisturbed and disturbed forest. *Cinnyricinclus femoralis* was recorded in undisturbed and disturbed forest.

(iii) Afrotropical highlands biome restricted species

A total of 40 ATHB species were recorded in this study (Appendix 8), with a total abundance of 2,647. This constitutes 57 % of all ATHB species and 74 % of ATHB species ever recorded in Mount Kenya. Observed and estimated species richness alongside abundance across each forest type are shown in Table 5.7.

Table 5.7: Observed (S(obs)) and estimated [(S(est)) based on Chao 1 and S(est) extrapolated)] species richness, total abundance, mean species richness and abundance per point count within undisturbed, disturbed and plantation forest types for Afrotropical highlands biome (ATHB) restricted species in Mount Kenya forest reserve. The values for species richness and abundance per point count are mean ± se. The values for estimated richness (S(est)) are mean ± 95 % CI of randomizations (100 randomizations) in each case. Superscripts next to values of S(obs) and abundance with no common letters denote significant difference of S(obs) and abundance of ATHB species across forest types, based on GLMs (p < 0.05). Means of species richness and abundance per point count with no superscript letters in common denote a post-hoc Least Significant Difference (LSD), which is significant between paired forest types (p < 0.05). Sample coverage = measure of sample completeness (maximum possible value of 1). % Change in richness = % change between S(obs) and S(est) Chao 1 richness (i.e., (increase/S_{obs}) *100). N is number of point count stations. Abun. = abundance.

Forest types	N	$S_{(obs)}$	Abun.	Richness per point count ± se	Abundance per point count ± se	S(est) chao 1	S(est) extrapolated	Sample coverage	% Increase (richness)
Undisturbed	75	29 [a]	748 [a]	7.99 ± 0.33 [a]	15.28 ± 0.45 [a]	31.00 ± 2.04	31.00 ± 0.00	1.00	0
Disturbed	75	31 [b]	1406 [b]	6.91 ± 0.30 [b]	13.44 ± 0.42 [b]	31.33 ± 5.94	31.13 ± 3.96	0.99	1.1
Plantation	40	22 [c]	493 [c]	6.00 ± 0.39 [b]	12.33 ± 0.56 [b]	23.00 ± 1.81	23.86 ± 2.46	0.99	4.5

The species accumulation and extrapolation curves of ATHB species approached asymptote in all three forest types (Figure 5.4). It also had a satisfactory sample coverage of ≥ 99 % (Table 5.7), with less than 5% difference between S(est) and S(obs) species richness for all forest types (Table 5.7). Therefore, species richness and abundance could be compared using S(obs) across forest types.
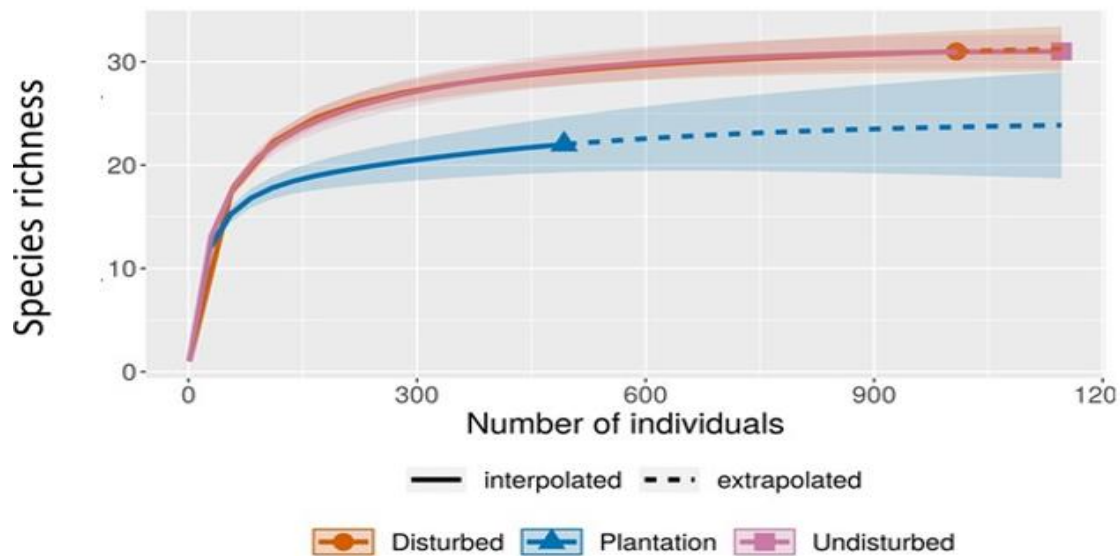
Figure 5.4: Species richness individual-based rarefaction and extrapolation curves for Afrotropical highlands biome restricted species in undisturbed, disturbed and plantation forest types, showing approach to asymptote.

Both bird species richness and abundance of ATHB were significantly different across forest types respectively (Wald $\chi^2 = 15.31$, df = 2, p < 0.001; Wald $\chi^2 = 18.45$, df = 2, p < 0.001), with highest species richness and abundance in disturbed forest type (Table 5.7). Species richness and abundance per point counts differed significantly across forest types but only pairwise differences between undisturbed and plantation, and between disturbed and undisturbed differed significantly with p < 0.05 (Table 5.7). Undisturbed forest had both the highest species richness and abundance per point count of ATHB, while eucalyptus plantation had the least (Table 5.7).

5.3.2 Prediction of bird species richness and abundance by habitat characteristics across forest types

Tables 5.8-5.10 shows statistically significant predictions of habitat characteristic variables grouped into vegetation structure, human disturbance and slope, altitude, and distance from forest edge on species richness and abundance response variables. The response variables are species richness and abundance for overall species, forest dependency, dietary guilds, and Afrotropical highlands biome restricted species (ATHB).

a) Prediction of vegetation structure variables

As shown in tables 5.8, canopy cover positively predicted forest specialists'(FS's) species richness and abundance, and species richness of frugivores, insectivores and ATHB species. However, it negatively predicted forest visitors' (FV's), non-forests' (NF's), granivores' and omnivores' species richness and abundance. % Vegetation cover at 0-1 m positively predicted forest generalists' (FG's) and insectivore's species richness, while % vegetation cover at 1-3 m positively predicted carnivores' species richness and abundance, and omnivores species richness. % Vegetation cover above 8 m negatively predicted species richness and abundance of nectarivores.

Number of saplings positively predicted frugivores species richness. However, it negatively predicted species richness and abundance of NFs, carnivores, and omnivorous birds. Tree diameter at breast height (DBH) positively predicted NF bird species abundance while negatively predicted frugivores species richness (Table 5.8). Tree heights positively predicted granivores' species richness, and canopy heights at > 20 m positively predicted granivores' species richness and abundance (Table 5.8). Litter depth negatively predicted nectarivores' abundance, while number of snags negatively predicted omnivores' species richness. Low canopy height at 0-10 m negatively predicted carnivore species abundance and frugivores species richness, while tree distance from centre of point counts negatively predicted frugivores species richness (Table 5.8). Foliage height diversity (FHD) negatively predicted omnivores' species abundance.

Table 5.8: Vegetation structure predictor variables that predicted positively/negatively the response variables. B = the regression slope, S.E. = standard error, and p = significance value (p-value).

Predictor Variable	Response variable	B	S. E	p
% Canopy cover	Forest specialists bird species richness	0.023	0.0056	< 0.001
	Forest specialists bird species abundance	0.022	0.0053	< 0.001
	Forest visitors bird species richness	-0.012	0.0061	0.047
	Forest visitors bird species abundance	-0.015	0.0059	0.012
	Non-forest bird species richness	-0.023	0.0088	0.008
	Non-forest bird species abundance	-0.021	0.0081	0.009
	Frugivore bird species richness	0.008	0.0025	0.001
	Granivore bird species richness	-0.013	0.0042	< 0.001
	Granivores bird species abundance	-0.016	0.0064	0.010
	Insectivores bird species richness	0.004	0.0017	0.025
	Omnivore bird species richness	-0.015	0.0064	0.021
	Omnivore bird species abundance	-0.025	0.0091	0.007
	ATHB bird species richness	0.007	0.0021	< 0.000
% Vegetation cover at 0-1m	Forest generalist species richness	0.005	0.0022	0.044
	Insectivores bird species richness	0.006	0.0019	0.001
% Vegetation cover at 1-3 m	Carnivore bird species richness	0.037	0.0189	0.050
	Carnivore bird species abundance	0.035	0.0178	0.048
	Omnivore bird species richness	0.022	0.0102	0.033
% Vegetation cover at > 8 m	Nectarivore bird species richness	-0.038	0.0122	0.002
	Nectarivore bird species abundance	-0.043	0.0133	0.001
Number of saplings	Non-forest bird species richness	-0.093	0.0379	0.013
	Non-forest bird species abundance	-0.137	0.0346	0.001
	Carnivore bird species richness	-0.09	0.0453	0.047
	Carnivore bird species abundance	-0.107	0.4301	0.012
	Frugivore bird species richness	0.029	0.0076	< 0.001
	Omnivore bird species richness	-0.073	0.0301	0.016
	Omnivore bird species abundance	-0.083	0.2254	0.019
Tree DBH (cm)	Non-forest bird species abundance	0.044	0.0219	0.046
	Frugivore bird species richness	-0.007	0.0033	0.038
Tree heights	Granivore bird species richness	0.035	0.0170	0.040
High canopy height at > 20 m	Granivore bird species richness	0.018	0.0091	0.044
	Granivores bird species abundance	0.028	0.0121	0.020
Litter depth	Nectarivore bird species abundance	-2.487	0.8536	0.004
Number of snags	Omnivore bird species richness	-0.361	0.1711	0.035
Low canopy height at 0-10 m	Carnivore bird species abundance	-0.281	0.1185	0.018
	Frugivore bird species richness	-0.056	0.0268	0.038
Tree distance from center of point counts	Frugivore bird species richness	-0.060	0.0284	0.034
FHD	Omnivore bird species abundance	-5.676	2.4897	0.023

ATHB = Afrotropical highlands biome restricted species, DBH = diameter at breast height, FHD = foliage height diversity

b) Predictions of human disturbance variables

Number of cut trees positively predicted FVs' species richness and NF birds species richness and abundance (Table 5.9). However, it negatively predicted overall species richness, forest specialists' species richness and abundance, and frugivores' species richness. Number of fallen trees positively predicted species richness and abundance for FS and frugivores. It however, negatively predicted species richness and abundance for FV and NF birds. It also negatively predicted species abundance for nectarivores and omnivores (Table 5.9). Number of trails positively predicted species richness and abundance for NF birds and granivore species (Table 5.9).

Table 5.9: Human disturbance predictor variables with response variables that predicted positively/negatively the response variables. B = regression slope, S.E = Standard error, and p = significance value (p-value).

Predictor variable	Response variable	B	S. E	p
Number of cut trees	Overall species richness	-0.035	0.0167	0.036
	Forest specialists bird species richness	-0.141	0.0694	0.042
	Forest specialists bird species abundance	-0.128	0.0652	0.047
	Forest visitors bird species richness	0.149	0.0758	0.046
	Non-forest bird species richness	0.246	0.0972	0.011
	Non-forest bird species abundance	0.322	0.0916	< 0.001
	Frugivore bird species richness	-0.093	0.0291	< 0.001
Number of fallen trees	Forest specialists bird species richness	0.219	0.0875	0.012
	Forest specialists bird species abundance	0.204	0.0847	0.016
	Forest visitors bird species richness	-0.247	0.0913	0.007
	Forest visitors bird species abundance	-0.304	0.0853	< 0.001
	Non-forest bird species richness	-0.550	0.1289	< 0.001
	Non-forest bird species abundance	-0.732	0.1146	< 0.001
	Frugivore bird species richness	0.178	0.0331	< 0.001
	Frugivores bird species abundance	0.219	0.0893	0.014
	Nectarivore bird species abundance	-0.268	0.1030	0.009
	Omnivore bird species abundance	-0.646	0.1231	< 0.001
Number of trails	Non-forest bird species richness	0.257	0.1063	0.015
	Non-forest bird species abundance	0.243	0.0921	0.008
	Granivore bird species richness	0.164	0.0686	0.017
	Granivores bird species abundance	0.252	0.0857	0.003

c) Predictions of slope, altitude, and distance from forest edge.

Slope negatively predicted omnivores' species abundance, while altitude negatively predicted overall species richness, and species richness and abundance for frugivores, and

granivores. However, altitude positively predicted species richness of insectivores and ATHB species, and species abundance of omnivores (Table 5.10). Distance from forest edge on the other hand positively predicted species richness of FS, nectarivores, and ATHB species. However, distance from forest edge negatively predicted species richness and abundance for FV, NF birds, granivores, and omnivores. It also negatively predicted nectarivores' species abundance (Table 5.10).

Table 5.10: Slope, altitude, and distance from forest edge predictor variables that predicted positively or negatively the response variables. B shows the regression slope, S.E. = Standard error, and p = significant value (p-value).

Predictor variables	Response variable	B	S. E	p
Slope	Omnivore bird species abundance	-0.034	0.0170	0.044
Altitude	Overall species richness	-0.994	0.2880	0.001
	Frugivore bird species richness	-0.001	0.0001	< 0.001
	Frugivores bird species abundance	-0.001	0.0003	< 0.001
	Granivore bird species richness	-3.342	1.0969	0.002
	Granivores bird species abundance	-0.001	0.0003	0.037
	Insectivores bird species richness	0.000	0.0079	0.022
	Omnivore bird species abundance	0.001	0.0004	0.024
	ATHB bird species richness	1.080	0.4994	0.031
Distance of point counts from forest edge	Forest specialist's species richness	0.616	0.2175	0.005
	Forest visitor's species richness	-1.002	0.2458	0.005
	Forest visitor's species abundance	-0.974	0.2354	< 0.001
	Non-forest bird species richness	-1.623	0.3912	< 0.001
	Non-forest bird species abundance	-1.831	0.3714	< 0.001
	Granivore bird species richness	-0.628	0.1749	< 0.001
	Granivores bird species abundance	-0.995	0.2369	< 0.001
	Nectarivore bird species richness	< 0.000	0.0048	0.009
	Nectarivore bird species abundance	-0.384	0.1638	0.019
	Omnivore bird species richness	-0.514	0.2520	0.041
	Omnivore bird species abundance	-0.704	0.3303	0.033
	ATHB bird species richness	0.162	0.0765	0.035

ATHB = Afrotropical highlands biome restricted species

5.4 Discussion

Forest birds' species richness and abundance may increase or reduce depending on the quality of forest habitats occupied (Tu et al., 2020). But degradation of natural forest structure through local human actions, and emergence of exotic plantation forests, potentially affects the quality of the forests and creates new habitats respectively for birds.

This can affect avian species richness and abundance on both local and regional scale (e.g., Stratford and Şekercioğlu, 2015; Bett et al., 2017). This study is a response to the need to understand the dynamic of birds to changing forest types created by human degradation of forests and change in associated habitat characteristics (including emerging exotic plantation forests), in Afrotropical montane forest areas (Rurangwa et al., 2021). This is to fill existing knowledge gaps and to inform appropriate forest management and conservation actions and for successful implementation of existing policy needs.

5.4.1 Effects of forest types on bird species richness and abundance

The finding that species richness and abundance of overall species, forest specialists, forest visitors, non-forest birds, ATHB species, carnivores, frugivores, granivores, and species abundance of nectarivores, are all significantly different across forest types shows that each forest type has a unique contribution to these groups of birds. Forest types however did not show any significant differences for species richness and abundance for forest generalists and insectivores, and species richness for nectarivores. Lack of statistically significant difference of insectivores' richness and abundance across forest types (Table 5.6) is contrary to what some researchers have found elsewhere (e.g., Şekercioğlu 2002a, Şekercioğlu et al., 2002b). Mahiga et al. (2019) found insectivore dietary guild to be significantly affected by forest types in the comparatively drier western part of Mount Kenya forest reserve. But the current findings agree with the findings of Gove et al. (2013) in Ethiopia's tropical montane environment. Lack of statistical differences across forest types for this guild in this study shows that all forest types are equally important for this guild and partly because of failure to distinguish various divisions of foraging strategies of insectivores, such as aerial, understory, terrestrial, or arboreal (Peh et al., 2015). These groupings may differ per forest types depending on the availability of food materials and sources (Johnston and Holberton, 2009). But considering them as one group may fail to reveal the differences across forest types.

Among the species of conservation concern, the only three of the eight KMEB species (i.e., 37.5 %) recorded in the current study (i.e., *Cisticola hunteri* (forest generalist),

Poeoptera kenricki (forest specialists) and *Cinnyricinclus femoralis* (forest specialist)), were the species that were within the range of the current study. The occurrence range of the rest was either beyond the confines of the study or were non-forest birds. These include *Turdoides hindei*, a non-forest bird confined to fringes of cultivation and river valleys (Zimmerman et al., 1999) and *Cisticola aberdare, Euplectes jacksoni*, and *Macronyx sharpei* that are mainly high elevation grassland birds and thus cannot be expected in the foothill and lowland montane forests. There was only one threatened species recorded (Abbott's starling (*Cinnyricinclus femorali*) (Vulnerable)) in this study. Therefore, the presence of this vulnerable species, together with the more than a third of KMEB species (all true forest birds) shows the conservation importance of Mount Kenya forest types for these species.

The 40 species of ATHB recorded in this study (out of 54 (i.e., 74 %) species ever recorded in Mount Kenya), with 29 species in undisturbed and 31 species in disturbed forest types (Table 5.7), demonstrate the importance of natural forest for the conservation of ATHB in Mount Kenya forest reserve. There were 22 species recorded in eucalyptus plantation forest (i.e., 55 % of the overall recorded ATHB species). Although this probably may mean that the eucalyptus plantations are potentially suitable to host sizeable number of bird species, the species in the eucalyptus plantation are mostly forest visitors and non-forest birds (See Chapter 6, section 6.3.1, and Figure 6.1 (b) and (d)). These species may be using the plantation for a short time refuge or for temporary use as an escape from, or in addition to the surrounding none forest farms and settlements. This therefore shows that eucalyptus plantation forest is just useful to an extent of only contributing temporarily to none-true forest birds of ATHB group in Mount Kenya Afromontane forests. From these results, it is possible that all forest types in Mount Kenya contribute to a greater or lesser extent, to the conservation of ATHB species, particularly ATBH forest specialist (17 species) and generalists (18 species), relative to forest visitors (5 species) (Appendix 8). These (specialist and generalist birds) are true forest birds that likely have strong relationship with natural forests (Morante-Filho et al., 2015).

(a) Species richness and abundance in undisturbed forest type

There were lowest number of overall bird species richness and abundance per point count in undisturbed forest than any other forest type. This was unexpected, but it could be because of the occurrence of few species that are more specialised and adapted to intact forests than those widespread and adapted to disturbed or/and eucalyptus plantation forests. Undisturbed forest is (expectedly) avoided by non-forest (NF) and forest visitors (FV) (which could be forming additional species in disturbed and eucalyptus plantation). For example, among true forest birds (forest specialists (FS) and generalists (FG)) (Bennun et al., 1996), FS's richness and abundance were significantly highest in undisturbed forest. It can be argued that this is expected for FS in undisturbed forest, where they tend to use it predominantly to obtain the needed specialised resources for their survival, and therefore may persist in undisturbed forest. High species richness and abundance of FS in undisturbed forest emphasise the irreplaceable importance of undisturbed natural forest for forest specialist birds in Mount Kenya Afromontane. This has also been similarly found in the neighbouring forests in the region e.g., Taita hills forest (Mulwa et al., 2021), Western part of Mount Kenya forest reserve (Mahiga et al., 2019), Wondo Genet Forest in Ethiopia (Girma et al., 2017), Dry evergreen Afromontane forests of Bale Mountains, Ethiopia (Asefa et al., 2017), four forest patches in Tanzania (Modest and Hassan, 2016) and Kakamega forest (Farwig et al., 2008).

FGs' species richness and mean species richness per point count did not differ significantly across forest types in this study (Table 5.5), probably because generalists tend to show no preference to a particular forest type (Bennun et al., 1996). However, forest types had statistically significant effect on FG's abundance and on mean species abundance per point count, probably due to the difference in availability of resources to individual birds in different forest types or due to intraspecific or interspecific competition among generalist's species (e.g., Tarjuelo et al., 2017). But contrary to this finding, FGs generally declined from natural forests to exotic plantations in Taita hills (Mulwa et al., 2021), western part of Mount Kenya forest reserve (Mahiga et al., 2019), and Kakamega forest (Farwig et al., 2008).

On dietary guilds, frugivores as expected had high species richness in undisturbed forest, likely this forest type acting as an important source of most intact fruiting vegetation and fruits availability. Elsewhere, primarily frugivores forest specialists' birds have been similarly found to occur in less disturbed habitats in tropical forests (e.g., Mulwa et al., 2021; Newbold et al., 2013). Undisturbed forest therefore is indispensable for frugivores forest birds in Afromontane forests, and if undisturbed forest is maintained undisturbed, they frugivores existence can thus be able to maintain the forests through seed dispersal (Lehouck et al., 2009; Mahiga et al., 2019; Costa et al., 2022). This is possible because the existing wilderness forest area/activity zone (e.g., Figure 3.5) in Mount Kenya forest can maintain the capacity for frugivorous birds to play a role in restoring neighbouring forest types, including eucalyptus plantations with native tree species. Eucalyptus plantation act as a sink for dispersed seeds and regeneration can be achieved, contributing to a landscape scale restoration of native forest trees (e.g., Amazonas et al., 2018).

(b) Species richness and abundance in disturbed forest type

Disturbed forest, contrary to expectation, attracted more overall bird species richness and bird abundance relative to undisturbed forest. This is probably so because the level of anthropogenic disturbance in disturbed forest may have created more habitat niches through vegetation structure modification (Bentsi-Enchill et al., 2022). Local anthropogenic disturbance may have also opened more opportunities for more invasive species to access the forest type, and probably benefitting it with more foliage related habitat variations (Morris, 2010). In Chapter 4 section 4.3.2 in this study, it is shown that foliage height diversity (FHD) has highest indicator value in disturbed forest type than other forest types, and it is one of the habitat characteristics that characterise disturbed forest rather than undisturbed forest. The level of habitat modification through disturbance (i.e., high percentage vegetation cover at 1-5m heights, high number of fallen trees, and number of saplings that characterised disturbed forest type in Chapter 4), could have led to foliage structure heterogeneity in disturbed forest. In addition, from researcher's personal observation in the field during this study, foliage vegetation was observed to be locally removed to get spaces for hive placement (see Chapter 4, Figure 4.4 (a)), as livestock feeds (Chapter 4, Figure 4.1 (b)), and as medicinal herbs, hence

probably affecting the way foliage structure is arranged both vertically and horizontally. Potentially also, open canopies in disturbed forest type may allow variability in light and temperature leading to structurally diverse mid, understory and forest floor (also see Khanaposhtani et al., 2012), thus encouraging establishment of high variability of percentage vegetation cover at heights between 1-5 m of the forest. For instance, Mulwa et al. (2021) reported that bird density and species richness increased with vegetation structural diversity in western Kenya. Similar finding of high species richness and abundance in modified sites was found in Afromontane forests in Ethiopia (e.g., Asefa et al., 2017; Leaver et al., 2019), Nensebo moist Afromontane forest in south-eastern Ethiopia (see Jemal et al., 2020), and in western Kenya (Mulwa et al., 2021). But modification in these habitats (except in Asefa et al., 2017 and Leaver et al., 2019) was mainly of agroforests systems and farmlands rather than being disturbed protected forest.

It is also important to note that despite local disturbance, the disturbed forest may still retain some features and characteristics of undisturbed forest, and these characteristics still attracts specialised birds and those adapted to intact forests in tropical Afromontane forests. This therefore reiterate the possibility of disturbed forest type to having high number of species richness and abundance, including forest specialists and generalists, with almost comparable species richness with undisturbed forest (Table 5.5).

In another perspective, the level of local disturbance in disturbed forest type could be intermediate between disturbance at exotic plantation forests (i.e., owing to silvicultural practices in plantations hence more cut trees and human trails, see Chapter 4 section 4.3.2) and undisturbed forest type. If this assumption holds, then the finding of more species richness and abundance in disturbed forest type in this study support the prediction of intermediate disturbance hypothesis, that local species diversity is at maximum at an intermediate level of disturbance (Connell, 1978; Wilkinson, 1999; Roxburgh et al., 2004). The disturbed forest type is likely not to be highly disturbed to warrant a loss of species or individuals, but rather it has resulted to attainment of intensities of disturbance that give maximum species, conforming to the predicted bell-shaped curve as suggested under intermediate disturbance hypothesis (e.g., Townsend et al., 1997; Wilkinson, 1999; Bongers et al., 2009). It can be concluded therefore, that the level of local disturbance in

Mount Kenya Afromontane forest might have created more niches for diverse birds for both richness and abundance.

Among dietary guilds, contrary to prediction of this chapter, frugivores had high species abundance in disturbed forest in this study. This probably demonstrates that most individual frugivore birds can also tolerate some habitat disturbances, or disturbance have opened more opportunities for fruit trees and fruits establishment that accommodate many individual birds. This agrees with arguments by Şekercioḡlu (2012) that certain frugivorous bird species may successfully forage in forests logged at intermediate intensities, where food is more abundant, while still nest in primary forests. Also, contrary to the expectation, species abundance for nectarivores were significantly highest in the disturbed forest. It was expected that due to less/no disturbance in undisturbed forest, and possible presence of variety of flowering plants, undisturbed forest could hold most species and abundance of nectarivore species, but this was not supported in this study. Probably the disturbance, and greater foliage height diversity that characterised disturbed forest type could have made the site to have greater diversity of flowering plants parts, that probably have attracted nectarivores to forage in disturbed forest relative to undisturbed forests.

(c) Species richness and abundance in eucalyptus plantation forest type

The higher-than-expected overall bird species richness and species abundance per point count in eucalyptus plantations as compared to undisturbed forest type, could be because of plantations being near natural forests and can be acting as a refuge site, probably providing additional forest resources and conditions that also attract both forest birds and NF birds. For instance, from personal observation during the study, it was observed that within the spaces in eucalyptus plantations, native herbaceous and shrubby vegetation also grow extensively. These vegetations could be forming habitats that attracts forest birds, making bird species and their abundance at eucalyptus plantation to be more than expected. In addition, owing to comparable conditions of eucalyptus plantation with those of farmlands and other surrounding non-forest areas (i.e., following silvicultural practices), it is possible that the eucalyptus plantations have attracted NF birds from these

sites. It is not surprising therefore, that FVs and NFs' richness and abundance are significantly different across forest types, with high species richness and abundance in eucalyptus plantation forest (Table 5.5). These birds (NF and FV) are attracted to disturbed conditions in eucalyptus plantations as they do not normally depend on forests according to Bennun et al. (1996).

Eucalyptus plantation offers similar/comparable opportunities to FGs as does other forest types (i.e., not significant different across forest types). However, it recorded only eight (8) FS species (Table 5.5) (i.e., see Figure 5.5 (a) for a FS bird in a natural undisturbed forest and a similar bird at eucalyptus plantation in the study area). Probably the alien status of eucalyptus, and probably the chemical content in their leaves and barks (e.g., Hayat et al., 2015), may explain the low suitability of this plantation for specialised forest birds such as FSs. Although subject to investigations, the eight FS bird species unexpectedly recorded in eucalyptus plantation may be using it as a supplementary habitat or as a temporary refugia. The structural features and growth characteristics of eucalyptus plantations may limit the availability of suitable local environmental conditions and food resources (e.g., Calviño-Cancela et al., 2012; Calviño-Cancela, 2013; Goded et al., 2019) for more FS.

Figure 5.5: Mountain greenbul (*Andropadus nigriceps*) a forest specialists bird recorded in Mount Kenya forest reserve at (a) undisturbed forest type in Ruthumbi forest site on 10/02/2019 and (b) a similar bird at eucalyptus plantation in Ruthumbi eucalyptus plantation on 04/05/2019. Photo credit: Author.

For dietary guilds, granivores had highest species richness and abundance in eucalyptus plantations and lowest in undisturbed forest. The observed high species richness and abundance of granivores in eucalyptus plantations could be explained by the presence of grain-bearing feeding materials in the plantation. This is probably owing to more open canopies in the plantations (see Chapter 4), allowing more sunlight that encourages development of undergrowth that consist of grasses and other grain bearing plants, that may be preferred by granivore bird species. This is probably true given that Mulwa et al. (2021) and Ndang'ang'a et al. (2013) also recorded strong positive relationship of granivore species richness and increased cover of seed resources on cultivation and fallow lands in the highlands cultivated landscape in Kenya. Pulliam and Brand (1975) also noted that granivores feeds on grass and forb seeds gleaned from the soil or from weedy annual plants.

The significantly highest nectarivore abundance per point count in eucalyptus plantation forest than any other forest type could be because of, in addition to likely easily visible/accessed flowers in the eucalyptus plantation, the observed native herbaceous and shrubby vegetation that grow extensively understory and at the edges of eucalyptus plantations possess nectar producing flowers that attract nectarivores. This findings on nectarivores agrees with that of Deikumah et al. (2017) who found that monoculture plantations favored nectarivore birds in a forest-agriculture landscape in Ghana. Calvina-Cancela (2013) also found that nectarivores were mainly associated with shrubs at young ages and with canopies at older ages in eucalyptus plantation in Galicia, northwest Spain.

5.4.2 Predictions of habitat characteristics on bird species richness and abundance in Mount Kenya forest reserve

a) Vegetation structure predictors

Habitat characteristics related to cover is mainly responsible for shaping bird community response to the predictors (Söderström et al., 2001; Heikkinen et al., 2004). Generally, it was found in this study that bird communities are strongly structured by vegetation characteristics mainly related to forest complexity. Vegetation structure predictors related to undisturbed forest (i.e., canopy cover, % vegetation cover at > 8 m, and number of

snags) (e.g., Chapter 4, section 4.3.2) significantly and positively predicted species richness and/or abundances for species that require undisturbed forest conditions. These included FS, frugivores, insectivores and Afrotropical highlands biome restricted species (ATHB). However, similar predictors significantly and negatively predicted bird species richness and/or abundance of FV's, NF's, granivores, carnivores, nectarivores and omnivores. The richness and/or abundance of the latter was significantly and positively predicted by vegetation structural characteristics related to open canopies i.e., in disturbed forest (e.g., canopy height at > 20 m) and in eucalyptus plantation (e.g., tree heights) (Chapter 4 (and see Table 4.7). These vegetation structural predictors relate to either simple forest (i.e., disturbed forest when coverage percentage of these characteristic are less after being removed through disturbance) (Alroy, 2017; Hart and Kleinman, 2018; Rajpar, 2018) or complex forest (undisturbed forest, when forest is intact and coverage is high) (Mishra et al., 2004; Wang and Cochrane, 2005; Baker and Spracklen, 2019).

But local forest disturbances such as selective logging can open/simplify forest canopies (e.g., Pfeifer et al., 2012 (a); Pfeifer et al., 2016). This can consequently encourage FV, NF, granivores, carnivores, nectarivores, and omnivores while discouraging FS, insectivores, and frugivores. On the other hand, establishment of eucalyptus plantation have attracted FV, NF, omnivores and granivores to it, while discouraging frugivores. For instance, the tall trees, together with greater distances from one tree to the other in eucalyptus plantation could be responsible for more open canopies that encourages undergrowth vegetation with grasses that attracts granivores and omnivores. Omnivores could also be attracted to disturbed sites owing to their lower diet specialisation and could be using it as additional source of food. For carnivore birds (significantly and positively predicted by vegetation cover at 1-3 m), they can prefer open disturbed natural forests to hunt for food, i.e., they can scan wider areas and have space to pursue prey. Nectarivore species (only negatively predicted by vegetation structures of undisturbed forest), could be using eucalyptus plantation sites as highlighted (section 5.4.1 c). Comparable to the findings of FV in this study, Mammides et al. (2015) also found that disturbance related structural changes (e.g., canopy openness) positively predicted species richness and abundance of FV birds in Kakamega forest, a remnant of a rainforest in Kenya.

Apart from canopy cover (associated with undisturbed forest), significantly and positively predicting insectivores' species richness, the same insectivores' richness was also predicted positively by vegetation cover at 0-1 m (that characterises eucalyptus plantation, see chapter 4, Table 4.7). The prediction of insectivores' richness by these features are contradicting, but insectivore guild could itself be composed of sub-sets of different insectivores with different foraging strategies. The insectivores attracted to eucalyptus plantation could be those associated with gleaning of invertebrates in the eucalyptus barks, or are high canopy gleaning insectivores (e.g., Ferger et al., 2014), that prefer taller trees than found in natural undisturbed forest, that could be understory insectivores (e.g., Şekercioḡlu et al., 2002 b).

Forest generalists' (FG's) species richness was positively predicted by high percentage vegetation cover at 0-1 m (characteristic of eucalyptus plantation) and at 1-3 m (characteristic of disturbed forest). This shows that they mainly closely associate with these forest type. But these being the only vegetation structure variables that predicted (negatively or positively) its richness, it empahsises the fact that this species is widespread utilising wider localities (e.g., Bennun et al., 1996). MacNally and Bennett (1997) argued that generalist species probably have a greater capacity to use range of microhabitats that occur in naturally heterogenous fragments whereas specialists may be restricted to only a portion of the fragment.

In natural forests (undisturbed and disturbed), high number of saplings and high tree diameter at breast height (DBH) measures are attributes of intact natural forests (Undisturbed, 2009; Borah et al., 2014). High number of saplings signify high chance of seed establishment and growth, while tree DBH shows age structure of forest stand (e.g., Ngoc Le et al., 2016). It also signifies less incidences of forest disturbance through selective logging (Morgan et al., 2019). But when vegetation characteristics of plantation forest are considered and compared together with that of natural forests, tree DBH fails to exhibit age structure of forest stand because growth characteristics of planted and natural forests are different. In this study, high tree DBH was found to show attributes of eucalyptus plantation forests, where increase in tree DBH predicted increase in NF birds' richness and abundance. This is separate from instances where tree DBH may positively

predict true forest birds (e.g., forest specialists) (see Deikumah et al., 2017), when only doing comparisons within natural forests. The forest establishment through saplings that negatively predicted NF's richness and abundance, is expected given that natural forest sites may have high potential to provide suitable environment for seeds establishment and growth than plantation areas (e.g., Mbora et al., 2009). NF's richness and abundance are therefore expected to be found more in the eucalyptus plantation forests, and probably in highly disturbed natural forests in Mount Kenya forest reserve. Studies have found greater tree seedling establishments in natural (i.e., undisturbed forests) areas with decaying woody plants, closed-canopy areas and areas with smaller forest gap sizes not so much exposed to direct solar radiation (Gray and Spies, 1997; Alvarez-Aquino et al., 2004; Mbora et al., 2009; Brodersen et al., 2019). Therefore, the direction of relationship of the number of sapling and tree DBH in this study, may correctly predict the existence of birds, for example NF birds in highly disturbed forest areas and in eucalyptus plantations.

In general, since vegetation provide structural elements for birds through provision of shelter, nesting sites and foraging substrate (Ferger et al., 2014), these elements seem to distribute birds according to their functional traits (forest dependency and dietary) across forest types in this study. For example, fruiting trees domiciled within natural forest, especially undisturbed forest, attract frugivores species richness, while those in disturbed forest attracts more frugivore abundance. Emergence of new vegetation characteristics such as exceptional taller trees and high tree-to-tree distance (provided by eucalyptus trees) than in natural forests, accommodate bird species such as carnivores, nectarivores, granivores and omnivores. This is probably because more light penetration to forest floor encourages undergrowth of seeded grassy materials utilised by the birds, especially granivores and omnivores. It is also probable that taller trees in eucalyptus plantation than in natural forests, have attracted foraging guilds such as high canopy insect gleaners (see Willson, 1974). Open canopies and high vegetation percentage at 1-3 m height from forest floor could have attracted carnivores, probably these characteristics offer opportunities for successful hunting by carnivore birds. from these findings, conservationists and forest managers in Afromontane forests ecosystems, can be therefore informed that each forest type that has been created or has emerged because of human use, serves a particular role in biodiversity conservation, with exotic forests such as eucalyptus plantation, appearing

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to offer much less to overall forest biodiversity conservation. Management strategies should therefore be based on the specific conservation goals, but much should be considered on emerging forest types, and how they successfully contribute to biodiversity conservation.

(b) Human disturbance predictors

The results associated with number of cut trees shows that forest disturbance (through selective logging) have detrimental effect on species richness and abundance of FS and frugivores birds. It also opens the forests to species richness of FV and richness and abundance of NF and granivore birds in Mount Kenya forest reserve. It reiterates other findings for example Althof (2005) who found that logging destroys plant communities leading to lower FS in disturbed sites. Forest disturbance also reduced species richness and density of frugivore community in three East African tropical rainforests (Budongo and Mabira forests in Uganda, and Kakamega forest in Kenya) (e,g., Kirika et al., 2008). However, number of trails (that was found to characterise eucalyptus plantation in this study (Chapter 4, table 4.7)), positively predicted NF's and granivores' species richness and abundance. Since these species are non-forest birds, their richness and abundance largely do not associate with natural forests characteristics but are attracted to eucalyptus plantations. It is therefore possible that, in addition to agricultural and human settled lands, eucalyptus plantation in Mount Kenya provides NF and granivores with additional areas to occupy probably with seeded or grain bearing grasses and forbs for granivores (e.g., Lees and Peres, 2009; Şekercioḡlu, 2012). This result is supported by the findings of Osuri et al. (2020), that abundances of granivores increases in degraded and converted habitats relative to intact forest. It was expected that human trails can have negative prediction on species richness and/or abundance of forest dependent birds (FS or FG), but number of trails only had positive prediction on NF richness and abundance in this study. For example, Bötsch et al. (2018) had argued that human trails play a role in creating disturbed sites within the natural forests, and act as proxy to human presence. But it could be possible that the number of trails were not enough or large enough, or not frequented much to cause noticeable effects on the bird species traits of forest birds considered.

c) Slope, altitude, and distance from forest edge

Although there was not much difference in altitudinal range between forest types (Chapter 3, Table 3.3), altitude nevertheless significantly and negatively predicted species richness of overall bird species, and for frugivores, and granivores species richness and abundance. In this study, high altitudes mainly (but not always) coincided with sites of undisturbed forest type and relatively low altitude coincided (but not always) with disturbed forest type. Less species richness in high altitude follows a long-held pattern of decreasing of species richness monotonically with increasing altitude because of temperature related habitat productivity (e.g., Stevens, 1992; Rohde, 1992; Körner, 2007). It also agrees with findings of Ghimire et al. (2021) in Central Nepal Himalayas. Although not related directly with bird species richness and altitudinal differences, Cirimwami et al., 2019 found that a woody lifeform follows a monotonic decrease of species richness in Kahuzi-Biega National Park, an East African montane forest located in Democratic Republic of Congo. Assuming altitude has an indirect influence on overall species richness mediated by habitat characteristic or richness of woody lifeform, then it may lower species richness at relatively higher altitudes in Afromontane forests. But this is subject to further investigation to disentangle effects of habitat characteristics or lifeform differences and altitudinal differences to determine which are of greater importance in explaining the current overall species richness distribution in the study area.

Association of frugivores and low altitude (that coincided with sites of disturbed forest type in the study) is contrary to the prediction of frugivores by vegetation and disturbance characteristics so far, that predicts that frugivores associate with undisturbed forest (Section 5.4.1 (a)). However, since undisturbed forest types were not always in high altitudes, it can be assumed that some frugivores species were recorded in low altitudes areas but within undisturbed forest. It needs also be remembered that disturbed areas are not devoid of characteristics found in undisturbed forest, i.e., some sites in disturbed forest type may mirror conditions in undisturbed forest (Alroy, 2017). Also, some frugivores species may also tolerate some habitat disturbance in tropical forests (e.g., Gomes et al., 2008). Indeed, frugivore's species richness and abundance are positively predicted by number of fallen trees (characteristics of disturbed forest) (Chapter 4 section 4.3.1 and

4.3.2) in the current study. This therefore reiterate the possibility of some frugivores recorded in low altitude within disturbed forest types likely to be the frugivores which are disturbance tolerant. It is also possible that local disturbance in Afromontane forests may encourage diversity of fruiting vegetations in the current study, although this argument is not supported by the findings of Kirika et al. (2008) in three East African tropical rainforests. This therefore raises a need to determine specific factors that drives frugivore distribution in disturbed forest sites in Afromontane forests.

Distance from forest edge could be important in explaining the differences in occurrence of forest dependency and dietary bird groups. For example, distance from forest edge positively predicted species richness of FS and insectivores, and negatively predicted species richness and abundance of FV, NF, omnivores, granivores and nectarivores. These differences in forest dependency and dietary guilds can be explained as caused by high anthropogenic disturbance (especially that destroy FS's and insectivores' suitable habitats) near the edges, and that pushes FS and some insectivores to less disturbed forest sites. Mammides et al. (2015) and Menke et al. (2012) found similar results in Kakamega forest, where shorter distances to the forest edge had negative effect on FS, and FG except for FV, that can benefit from higher levels of disturbance. Long distances from forest edges could be indirectly preventing human activities from affecting the interior of forests, where humans are less likely to reach, and therefore enhancing existence of intact forest, FS, and some insectivore birds (probably those associated with the understory habitats).

For insectivores, there are somewhat contradicting predictions based on vegetation structural characteristics, altitude (with some linking insectivores to disturbed and eucalyptus plantation and others to undisturbed forest). It has been argued before (section 5.4.1) that there are various insectivore sub-groups not considered in this study such as understory insectivores, aerial gleaning, or foliage gleaning insectivores (Şekercioḡlu et al., 2002 b; Gove et al., 2013; Peh et al., 2015). Consideration of only combined insectivores might have masked the specific insectivore's response to prediction by habitat characteristics, and it is not surprising that this guild did not differ significantly across forest types (section 5.4.1). However, it has been found elsewhere that diversity of arthropods (food materials for insectivores) decreases with increasing elevation, and that

is expected to reflect a general decrease in species richness of insectivores with increasing elevation (see Guevara and Aviles, 2007). But contrary to this, insectivores' richness was positively predicted by altitude (Appendix 6 c) in this study. This is possibly in response to increase in food materials (arthropods or some other influencing factors) with altitude which is a selective force determining foraging behaviour and resources exploitation by birds (Sam et al., 2017).

Habitat characteristics (canopy cover, altitude, and distance from forest edge) are predicting ATHB species richness to be positively associated with undisturbed forest types. However, the composition of ATHB species also indicate that a sizeable number of species (e.g., 55 % of the recorded ATHB species) occupy exotic eucalyptus plantation (Table 5.7). But ATHB species are restricted range species (see Scharlemann et al., 2005), and are part of species assumed to have developed some adaptation to environmental conditions within their range (De Klerk et al., 2002; Polechová and Barton, 2015), although there is possibility of range-shifts owing to other factors such as climatic changes (Shay et al., 2022). The observation of more than half of the recorded ATHB restricted species on eucalyptus plantation, with none of the eucalyptus plantation habitat characteristics predicting its richness and/or abundance is interesting. Undisturbed forest type in the study was relatively at high altitude and was at between 3-4 km from forest edge, coinciding with where ATHB species richness are predicted to be. But eucalyptus plantation was at intermediate altitude with little difference in altitudinal range with undisturbed forest. The presence of ATHB in eucalyptus plantation could probably only act as a refuge or temporary site (since it is an exotic plant, probably not well adapted with local birds) but probably not mandatory for their survival as does undisturbed forest type. In other regions in the tropic, restricted range species have been found to be less in number (Marsden et al., 1997; Oostra et al., 2008), and found in both forests and non-forest areas. Some of the restricted-range species have been found to be at greater risks from extinction due to habitat change (Scharlemann et al., 2005), suggesting that they are more specialised to habitat characteristics.

In general, granivores, nectarivores and omnivore species richness and abundance all were positively predicted by vegetation structures and human disturbance characteristics

associated with disturbed and eucalyptus plantation forest types. It is not therefore surprising that they are negatively predicted by distance from forest edge and altitude. However, their predictions by altitude and distance from forest edge can be explained based on food resource availability, which changes with disturbance and altitude for these groups (e.g., Sam et al., 2017). For example, increase in disturbance towards the forest boundary may be encouraging availability of resources for granivores, nectarivores and omnivores, while doing opposite for some insectivores and frugivores (i.e., Mammides et al., 2015; Sam et al., 2017). From a conservation point of view, lack of, or reduced insectivores and frugivores closer to agricultural farms (near forest edges) may mean either there will be reduced pest control by insectivores, and reduced seed dispersal by frugivores in disturbed forests. This may hinder development of appropriate self-regulating ecosystems and forest regeneration processes needed.

For nectarivores species, although positively predicted by vegetation characteristics related to disturbed areas and eucalyptus plantations, its' richness is positively predicted by distance from forest edge, while its abundance is negatively predicted by distance from forest edge. This means the richness are associated with some distance away from edges, possibly undisturbed forest type in this study while abundance does the opposite. In fact, nectarivores' species richness did not differ significantly across forest types, and this probably contribute to the seemingly universal distribution of this guild across forest type. It therefore demonstrates its availability to offer pollination services across forest types.

Granivores richness and abundance, predicted to associate with disturbed and eucalyptus plantation forest types near forest edges, may also become agriculturally detrimental (acting as crop pests) dominating near forest boundary. It is therefore both beneficial for the agricultural lands nearby, and forest conservation's sake to conserve forest integrity right from boundaries but being aware of naturally occurring conflicts between conserved species and agricultural activities.

5.4.3 Management and conservation implications

Despite local anthropogenic disturbance assessed in this study and emergence of exotic eucalyptus plantation, all forest types in Mount Kenya forest reserve are important for

bird conservation and potentially to all other taxa. Each forest type contributes uniquely to the needs of birds (See section 6.3.1), with natural forests being particularly important to the specialised bird like forest specialists, FG, frugivore birds, Kenya mountains endemic birds (KMEB), ATHB and one vulnerable species, all mainly dependent on undisturbed forest. Most FS, KMEB, ATHB and the only threatened species (Abbott's starling (*Cinnyricinclus femorali*) (Vulnerable)) cannot be found in any other forest type apart from natural forests thus natural forests are irreplaceable for them. The only way to conserve such species is to maintain undisturbed and conserve disturbed forest sites for perpetual occurrence of these group of species in Mount Kenya Afromontane forest. This can be done through zonation of protected forest areas and setting aside a strictly protected core zone areas to maintain specialised undisturbed forest dependent species.

In this study, it is evident that there is high overall species richness, and number of ATHB species richness (i.e., out of 40 species recorded, 31 (78 %) species in disturbed compared to 29 species in undisturbed forest) are found in disturbed forest type than undisturbed and eucalyptus plantation (Table 5.7). It could be that the level of local disturbance in disturbed forest type in the study area is not ecologically destructive to all birds. The level of disturbance could only serve to increase the niches for birds, i.e., through foliage modification and therefore attracting more overall bird species richness and abundance and those of ATHB restricted range species. For example, the high impact of local anthropogenic disturbance is acting to increase vegetation covers at 1-5 m heights from forest floor, canopy heights, and generally FHD (see Chapter 4, Table 4.7). This makes the disturbed forest to have more species richness overall and those of ATHB group than expected. Ecologists and conservationists/forest managers have been interested to understand whether there is any evidence of species sensitivity to moderate habitat disturbance (e.g., Gray et al., 2007; Keinath et al., 2017). In addition, KFS (2010) and KWS (2010) have also been questioning the role of permitted anthropogenic disturbance as part of participatory forest management (PFM) in affecting forest biodiversity. This study has demonstrated that different groups of birds in Mount Kenya Afromontane forest are affected differently, but overall species richness and abundance is high with the current disturbance in disturbed forest. However, it is detrimental to species richness and abundance of most FS and some frugivores birds, while opening opportunities for species

richness of FV and richness and abundance of NF and granivore birds in Mount Kenya forest reserve. Loss of frugivores, i.e., through cut trees/selective logging may lead to negative impacts to the ecosystem services provided by frugivores in seed dispersal, risking a failure for appropriate forest regeneration. Managers and conservationists (i.e., KFS and KWS) should therefore be aware of the appropriate ways of conserving and maintaining more and specialised species such as frugivores and forest specialists, and to strategize on how to reduce of excessive local forest disturbance. In addition, the management should maintain appropriate habitat vegetation structural manipulation in natural forests, that enhances biodiversity rather than destroy.

The eucalyptus plantation on the other hand has introduced at least some probably useful habitat characteristics such as taller trees, dense undergrowth, and increase in tree-to-tree distances that have served to attract mostly species richness and/or abundance of some FG, FV, NF, omnivores, nectarivores, and granivore birds. It is therefore important for management and conservationist to note that, although the above is the case, the potential conservation values of eucalyptus plantation are not yet fully appreciated since the species richness and abundance of birds observed in the plantation (particularly for FS), could be just because of proximity of the plantation to natural forests, or the plantations are just acting as a mere refuge or temporary sites. It could just be providing additional sites for non-forest birds or for generalist birds, probably serving to reduce use pressure and competition in surrounding farmlands and other non-forest sites in Afromontane. Eucalyptus plantation habitats has not been yet proven beyond reasonable doubt in this study (or any other e.g., Jacoboski et al., 2016; Phifer et al., 2017; Goded et al., 2019), on its suitability for forest dependent birds. Therefore, any management and conservation strategies that can be designed to include eucalyptus plantation as key habitats for biodiversity conservation should take cognizant of the minimal contribution of eucalyptus to forest birds' conservation in Afromontane forests, but it can provide refuge and temporary sites for non-forest birds and forest generalists, especially granivores and omnivores, in addition to commercial use that are meant to serve. Harvesting and other eucalyptus use should then be implemented in a conservation friendly manner to the latter groups of birds in Afromontane sites and other areas.

Since eucalyptus plantation mainly associate with presence of granivore birds, the more granivores in eucalyptus plantation may play a destructive role in tropical Afromontane forests by removing or consuming any regenerating natural forest seedlings, shoots, or seeds (e.g., Wright, 2003; Terborgh et al., 2018) leading to slow, poor or no forest re-establishment. This possibility should be noted particularly in area that eucalyptus plantation borders farmlands, and any pest control measures designed should include how to control granivorous birds damaging crop lands. Similar consideration may be made for any possible human-wildlife conflicts arising from birds that are acting as crop pests particularly where eucalyptus plantation is near agricultural lands.

5.4.4. Conclusion

High species richness and abundance of overall species, ATHB restricted range species, and species abundance for FG, frugivores and nectarivore birds in Mount Kenya Afromontane forest is as a result of response of these bird groups to local disturbance within disturbed natural forests. This local disturbance is responsible also for encouraging some FV into disturbed forest, while discouraging most FS richness and abundance, and frugivore richness. The disturbance meted on this forest type on habitat characteristics act to diversify the contribution of foliage, creating a variety of niches rather than destroying it for different bird groups. This is encouraged by participatory forest management approach which, with its level of allowable disturbance (e.g., firewood collection, livestock feed harvesting, beehive placement etc) is more ecologically appropriate than previously thought (e.g., Kareiva et al., 2007; Bradshaw et al., 2009; Nasi and Frost, 2009; Chisika and Yeom, 2021) or predicted in this study. Undisturbed forest serves to maintain forest habitat characteristics attributes, mainly based on forest complexity attributes that positively predict richness and abundance of forest specialists, ATHB restricted range species, and species richness of frugivore birds. Eucalyptus plantation on the other hand is highly disturbed through tree cutting and trails and have created new habitat characteristics with tallest trees, more open canopies and greater percentage of vegetation covering 0-1 m from forest floor (mainly composed of native herbaceous and shrub vegetation). These have positively predicted mainly richness and abundance of FV, NF, granivores, omnivores, and abundance of nectarivores bird species in eucalyptus

plantation. Eucalyptus plantation have therefore created possible temporary refuge or new sites for bird species from both forested and non-forested surrounding areas to Afromontane forests.

Chapter 6-Bird community composition across forest types in Afromontane forest, Mount Kenya

6.1 Introduction

Across the globe, no forest ecosystem is devoid of human impacts (Neuschulz et al., 2011), but these impacts vary depending on where humans and their different intensities of actions have reached (Popradit et al., 2015). For example, it has been found that for forests inside protected areas (i.e., inside forest reserves or national parks) only 56 % have high habitat integrity and only 40 % have high ecosystem integrity (Grantham et al., 2020), while the rest have undergone some form of anthropogenic modification. In tropical forests anthropogenic activities are the major cause of decline in biodiversity (Morris, 2010), causing disturbances that influence forest integrity and dynamics at local scales (Sapkota et al., 2009).

In the tropics, communities living within 5 km of the forest boundary make most use of the forest resources, and it is the peripheral forest areas (generally within 3 km inside the forest boundary) that are mostly heavily used (Wass, 1995; Beche et al., 2022). Moreover, studies have found that about 20 % of the remaining tropical forests exist at less than 100 m from the forest edge (Haddad et al., 2015; Brinck et al., 2017). Therefore, due to varying anthropogenic pressures exerted at different parts of protected forests, there exist different forest types that reflect disturbance regimes. These forest types range from disturbed forests near the edges to undisturbed forests toward forest interiors in the natural forests. In addition, forest managers usually restore degraded natural forests sites, or do afforestation in formerly treeless sites within protected areas, using exotic trees in form of plantations (e.g., Lamb and Gilmour, 2003). This creates additional forest type to the existing natural forest types (that differ based on disturbance intensity). Therefore, in tropical protected areas there is existence of areas with high disturbance and those with low disturbance across natural forests, and also there are areas occupied by exotic plantations. Sites near the forest boundaries that are easily accessed are highly disturbed, while the sites that are difficult to access based on either distance, legal permission, topography, safety, and climatic conditions can remain largely undisturbed (Wass, 1995).

It has been noted that the species composition of animal community changes following forest disturbance (Carreño-Rocabado, 2012; Gray et al., 2018) and the conventional view is that after disturbance wildlife density and species richness are low (Green and Sanecki, 2006). Also, across the tropics the general trend exists of significant reduction in species composition from the complex natural forests to simplified monoculture habitats (Simamora et al., 2021). It is expected therefore that the existence of anthropogenic created forest types within the tropics, may create different biodiversity community composition distribution across the forest types, with highly disturbed areas having low diversity. This is because community composition in an ecosystem is driven by the quality of the ecosystem such as forest complexity, and changes in habitat characteristics through disturbance affect this quality (Latja et al., 2016; Batisteli et al., 2018).

Emergence of exotic plantation forests is likely to result into the existence of another different community compositions of organisms, whose individual components may, or may not be, a subset of those occurring in natural forests (Bowyer, 2006; Brockerhoff et al., 2008; Brockerhoff et al., 2013). There is a high possibility that exotic plantations, with its habitat features that may differ with that of natural forests, may attract species components from other forests and non-forest areas (Brockerhoff et al., 2008). Studies have reported that some wildlife species may lose out or benefit from newly created habitats features and conditions such as farmlands or plantations (Ranganathan et al., 2008). For example, some plantation forests are found to be almost devoid of wildlife/bird communities (e.g., Mulwa et al., 2021), while others find it suitable for some species as surrogate habitats (Calviño-Cancela, 2013; O'Callaghan, et al., 2017). Therefore, it is inconclusive as to which specific factors drive animal composition and distribution within the tropical forests that are influenced by local anthropogenic factors, across natural forests and emerging plantations.

Some studies have found similarities in bird species communities within different forest types in tropical forest ecosystems (e.g., Bael et al., 2013; Mulwa et al., 2021), and similarities in species richness and abundance in differently disturbed forest types (e.g., Shahabuddin and Kumar, 2006). For example, Mulwa et al. (2021) found that bird communities in small forest fragments were more like in agricultural land than in the near-

natural forest block in Taita Hill forest, Kenya. Therefore, it is evident that some changes in bird species composition and those of functional traits can be expected following human disturbance (e.g., Farwig et al., 2008; Neuschulz, et al., 2011). Forest specialists, for instance, have been shown to be more susceptible to human impact and thus might be replaced by generalist or forest visitor species in disturbed habitats (Peh et al., 2005; Farwig et al., 2008). Also, when different birds' dietary guilds are compared, frugivores, insectivores, and nectarivores are more associated with forests, while granivores bird species are more associated with agricultural lands (e.g., Ndang'ang'a et al., 2013). Although it is expected that forest birds and their forest dependency groups are more similar across natural forest types, than between natural forest and exotic plantations or other ecosystems, the community differences may still occur, reflecting the nature of forest types with regards to disturbance intensities and structural characteristics. Local human disturbance and expansion of exotic plantation is inherent in tropical montane forests of Africa and have generated existence of different forest types (Mbugua, 2000; Bekele, 2001; Chileshe, 2011). The resultant forest types are characterised by different habitat vegetational structural and disturbance characteristics highlighted in Chapter 4, yet it remains largely unknown to what extent these differences in forest types affect bird species composition in tropical Afromontane. The change in species composition may substantially alter ecological functions of ecosystems (Terborgh et al., 2008) and Afromontane forests are of no exception if forest characteristics change because of local anthropogenic actions and emergence of exotic plantations.

Different groups of birds, or individual bird species can characterise different forest types depending on how closely they associate with it (e.g., Asefa et al., 2017; Simamora et al., 2021; Yasin and Tekalign, 2022). They act as indicator species that are sensitive to change, and whose composition in each forest type may indicate certain environmental conditions (Lindenmayer et al., 2000). Some species may be shared among forest types, and others may be unique to a given forest type (Garcia et al., 1998). Shared species for example, may reflect their generality in habitat use, and less sensitivity to forest structural differences between the forests. Unique species to a forest type on the other hand, may reflect their fidelity to conditions and resources available in each forest type that are critical to their survival. These species may act as indicator species to a forest type where

they are highly sensitive to its change in quality. Unique species to a given habitat or forest type can be used as surrogate species, that reflect and signify the ecological conditions available within it (Githiru et al., 2007). To the best of current understanding, there are no studies to date that have looked at the unique or shared composition of birds across existing forest types in Afrotropic montane forests. There is also limited understanding of the conservation values, in terms of community composition of current emerging forest types or land use (resulting from widespread local anthropogenic actions in protected areas), and the exotic plantations (KFS, 2010; KWS, 2010). Following these, there is therefore a need to assess the current contribution of exotic plantations to community composition in Afromontane forests, as this plantation forms a substantial part of the Afromontane forest ecosystem and it is expanding. Establishment of plantation, together with existence of structurally different natural forests, could have led to possible re-arrangement of bird species and changes in community composition across these forest types. This possible re-arrangement of species, in terms of compositions, have not been investigated in Afrotropical montane forests. From the conservation and management point of view, information on specific factors that drives wildlife composition and distribution in tropical forests is needed to guide decision making, especially where there are widespread anthropogenic activities affecting natural forests (Dudley et al., 2008; KWS 2010).

The aim of this chapter was therefore to describe bird community composition and how it varies across three forest types (undisturbed, disturbed and eucalyptus plantation) in lower Afromontane forests of Mount Kenya. The chapter tries to answer the following questions:

(1) how are bird species compositions shared across the forest types, and are species composition unique to each forest type?

(2) how does bird community composition differ across forest types?

(3) what bird species are characteristics of each forest type?

It is predicted that shared species across forest types are widespread species, mostly forest generalist birds, and omnivores that are likely to depend on wider habitat resources and

conditions (wider niche). Unique species to each forest type are predicted to reflect the specific requirements of those birds in each forest type. It is predicted that forest birds' composition responds to different forest types depending on their close association with it (Barlow et al., 2007 (a)). For example, forest specialists, frugivores, insectivores and nectarivores are predicted to form assemblages within undisturbed forest while granivores and carnivores likely to form assemblage within eucalyptus plantation. Forest generalists are expected across all forest types (Neuschulz et al., 2011; Carlo and Morales, 2016), while forest visitors and non-forest bird composition are predicted to be found in disturbed and eucalyptus plantations respectively. Omnivores are predicted to be widespread because they can adjust their diet and can be found in different habitat types (e.g., Krimmel, 2011). Species of conservation concern such as Afrotropical highlands biome restricted species are expected to have greater composition within natural forests, and especially undisturbed forest (Wamiti et al., 2010), due to adaptation to it. Forest specialists are expected to characterise undisturbed habitats while forest visitors, and non-forests bird individuals are expected to characterise either disturbed and/or eucalyptus plantations.

6.2 Methods and data collection

In the eastern and southeastern foothills of Mount Kenya, three forest study sites were selected. Each site consisted of a disturbed forest (a section between the forest boundary and 1 km into the forest), undisturbed forest (a section between 3-4 km into the forest interior) and a eucalyptus plantation, situated within the designated Mount Kenya forest reserve protected area, and neighboring the natural protected forests. Study area information, sites, study design and data collection protocol are detailed in Chapter 3, section 3.2.1. It includes details of bird data collection (species identification and abundance) collected via point count surveys (Chapter 3, section 3.3, and sub-section 3.3.1).

6.2.1 Data management prior to analysis

The data which include species richness and abundance was used in this chapter to determine bird community composition across forest types. The detail of data

management prior to analysis is detailed in Chapter 3 section 3.3.2 (a) and Appendix 3 (a) and Appendix 3 (b). This includes how species richness and abundance was handled over the four survey seasons and over the two temporal data collection periods within each season.

6.2.2 Data analysis

The observed number of shared and unique birds to each of the forest types were determined and presented using Venn diagrams. Analysis of similarity (ANOSIM) was used to assess differences in bird species composition between forest types (Yasin and Tekalign, 2022). Global R values were used to determine the degree of similarity among forest types in ANOSIM; where the closer the value is to 1, the more dissimilar are the species composition. This was done by examining the Bray-Curtis similarity index from 9999 permutations in PAST Version 4.02 (Hammer et al., 2001; Hammer, 2020). The difference in bird species composition among forest types were further investigated and illustrated using non-metric multi-dimensional scaling (NMDS) using Bray-Curtis index in PAST v4.02. Plots of NMDS were applied as visual aids to interpret how species composition differed among forest types. The borderline connecting the outermost points for forest types in NMDS was used to delineate bird species composition for each forest type.

Similarity percentage analysis (SIMPER) was also conducted in PAST v4.02 to calculate the percentage contribution that each species made to the dissimilarity between bird assemblages of the forest types and to identify which species were contributing most to the differences (i.e., species that are characteristic of each forest type) between forest types (Asefa et al., 2017; Gumede et al., 2022; Yasin and Tekalign, 2022). The species that are characteristics to a given forest type are the ones with a high mean abundance in SIMPER in that forest type. Only species that contributed to the first 50 % of the cumulative percentage of dissimilarity were considered for dissimilarity comparison (e.g., Asefa et al., 2017). The contrast of species composition on ANOSIM and SIMPER was based on Bray-Curtis distance measure of dissimilarity metric (Hammer et al., 2001; Hammer, 2020). This is because, relative to other measures, Bray-Curtis is better at

handling the large proportion of zeroes (i.e., species absences) commonly found in ecological data sets, and this measure does not consider shared absences as being similar, which makes sense biologically (Schroeder and Jenkins, 2018).

6.3 Results

6.3.1. Shared and unique bird species composition across forest types

To illustrate the number of shared and unique species (only found in each circle) of all species combined, species were grouped into forest dependency, dietary guilds, and Afrotropical highlands biome species between different forest types, Venn diagrams were produced where different colours represent different forest types (Figure 6.1). The number of species shown in each circle represent the unique species per forest type while the number of species between two intersecting circles represent shared species between those two intersecting circles. The number of species within three intersecting circles represent shared species in three intersecting circles.
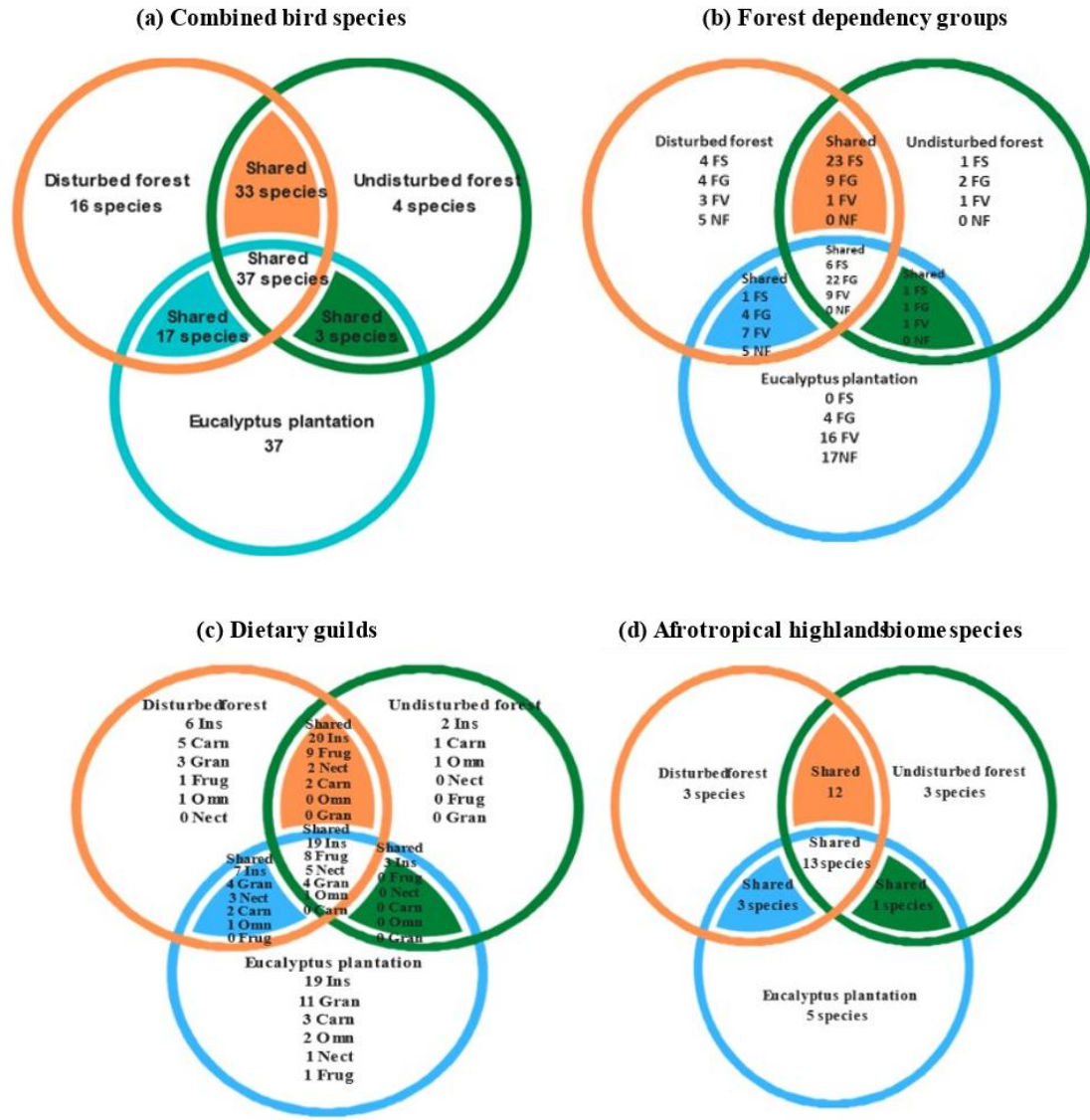
Figure 6.1: Shared and unique bird species for overall, forest dependency, dietary guilds, and Afrotropical highlands biomes across forest types. Totals of forest dependency in (b) and dietary guilds in (c) gives total numbers given in combined bird species in (a). Forest dependency (FS = forest specialists, FG = forest generalists, FV = forest visitors, NF = non-forest) and dietary guilds (Ins = insectivores, Frug = frugivores, Nect = nectarivores, Carn = carnivores, Omn = omnivores, Gran = granivores) are broken down to groups with numbers that add up to overall combined numbers.

Majority of overall bird species were shared across all forest types (i.e., 37 species or 25 % of all 147 recorded species in the study) (Figure 6.1 a). This shows existence of possible suitability of all forest types to most existing bird species in the study area, and possible complementarity among forest types on bird species composition. This is also reflected

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on the shared forest dependency assemblage and dietary guilds across all forest types, where unsurprisingly most of them are forest generalists (22 species), and insectivores (19 species) and frugivores (8 species) (Figure 6.1 c). In addition, all the forest types serve to accommodate most of the Afrotropical highlands' biome restricted species, where 13 out of 40 or 32.5 % are shared across all forest types (Figure 6.1 d).

Even in any of the paired forest types, the forests complement each other in species composition to some extent, but based on the nature of forest (i.e., if natural or exotic). For example, between undisturbed and disturbed forest types, they share majority of the overall bird species composition (i.e., 33 species) but between disturbed and eucalyptus plantation (both share 17 species), while undisturbed and eucalyptus plantation share only 3 species (Figure 6.1 a). On forest dependency, most shared species between undisturbed and disturbed forest types are true forest birds (FS = 23 species and FG = 9 species) (Figure 6.1 b). But there are few of these species group between pairs of disturbed/undisturbed and eucalyptus plantation, yet there is increase of shared FV and NF bird's species composition between disturbed and eucalyptus plantation (Figure 6.1 b). On dietary guilds, most shared species between undisturbed and disturbed forest types are insectivores and frugivores species, which reduces between disturbed and eucalyptus and further reduces between undisturbed and eucalyptus plantation. In addition, a third of ATHB species composition recorded (i.e., 12 out of 40) was shared between undisturbed and disturbed forest types (Figure 6.1 d). This is far more than what is shared between disturbed and eucalyptus plantation and those shared between undisturbed and eucalyptus plantation forest types.

The overall unique species number per forest type unexpectedly increased from undisturbed to plantation forest types (i.e., undisturbed = 4, disturbed = 16, plantation = 37) (Figure 6.1 a). However, as predicted, unique forest visitors (FV) and non-forest (NF) bird species numbers also increased from undisturbed to eucalyptus plantation forest type (i.e., undisturbed = 1 FV and 0 NF, disturbed = 3 FV and 5 NF; eucalyptus plantation = 16 FV and 17 NF respectively). It also has an unexpected pattern of increase in unique insectivore's species numbers from undisturbed to eucalyptus plantation (i.e., undisturbed = 2 species, disturbed = 6 species, and eucalyptus plantation = 19 species) (Figure 6.1 c),

as well as unique granivore species richness increase from undisturbed to eucalyptus plantation forest types (i.e., undisturbed = 0, disturbed = 3, eucalyptus plantation = 11). Furthermore, eucalyptus plantation exclusively accommodated 5 species belonging to ATHB (Figure 6.1 d, Table 6.3), a number which is unexpected, and more than found in disturbed and undisturbed forest type.

The table is presented below showing individual bird species composition unique to each forest type, with their reducing numbers from eucalyptus plantation (Table 6.1), disturbed forest type (Table 6.2) to undisturbed forest type (Table 6.3).

Table 6.1: Individual bird species that were unique to eucalyptus plantation forest type. FD= forest dependency, FS = forest specialist, FG = forest generalist, FV = forest visitor, DG = dietary guild, ATHB = Afrotropical highlands biome species, IUCN Cat. = International Union for Conservation of Nature red list categories, and Plant. = plantation forest.

Species	Scientific name	Family	FD	DG	IUCN Cat.	ATHB	Plant.
Abyssinian Crimsonwing	*Cryptospiza salvadorii*	Estrildidae	FG	Granivore	LC	✓	✓
Blackcap	*Sylvia atricapilla*	Sylviidae	FG	Insectivore	LC		✓
Grey wagtail	*Motacilla cinerea*	Motacillidae	FG	Insectivore	LC		✓
Montane Nightjar	*Caprimulgus poliocephalus*	Caprimulgidae	FG	Insectivore	LC	✓	✓
African Citril	*Serinus citrinelloides*	Fringillidae	FV	Granivore	LC		✓
African Harrier-hawk	*Polyboroides typus*	Accipitridae	FV	Carnivore	LC		✓
Baglafecht Weaver	*Ploceus baglafecht*	Ploceidae	FV	Omnivore	LC	✓	✓
Black-and-white Mannikin	*Spermestes bicolor*	Estrildidae	FV	Granivore	LC		✓
Brown-backed Scrub Robin	*Cercotrichas hartlaubi*	Muscicapidae	FV	Insectivore	LC		✓
Cardinal Woodpecker	*Dendropicos fuscescens*	Picidae	FV	Insectivore	LC		✓
Common Bulbul	*Pycnonotus barbatus*	Pycnonotidae	FV	Omnivore	LC		✓
Eastern Honeybird	*Prodotiscus zambesiae ellenbecki*	Indicatoridae	FV	Insectivore	LC		✓
Golden-winged Sunbird	*Nectarinia reichenowi*	Nectariniidae	FV	Nectarivore	LC	✓	✓
Grey-capped Warbler	*Eminia Lepida*	Cisticolidae	FV	Insectivore	LC		✓
Little Sparrowhawk	*Accipiter minullus*	Accipitridae	FV	Carnivore	LC		✓
Ring-necked Dove	*Streptopelia capicola*	Columbidae	FV	Granivore	LC		✓

Species	Scientific name	Family	FD	DG	IUCN Cat.	ATHB	Plant.
Spectacled Weaver	*Ploceus ocularis*	Ploceidae	FV	Insectivore	LC		✓
Streaky Seed-eater	*Serinus striolata*	Fringillidae	FV	Granivore	LC	✓	✓
Tree Pipit	*Anthus trivialis*	Motacillidae	FV	Insectivore	LC		✓
Yellow crowned canary	*Serinus canicollis flavivertex*	Fringillidae	FV	Granivore	LC		✓
African Cuckoo	*Cuculus gularis*	Cuculidae	NF	Insectivore	LC		✓
Augur Buzzard	*Buteo augur*	Accipitridae	NF	Carnivore	LC		✓
Brimstone Canary	*Crithagra sulphurata sharpii*	Fringillidae	NF	Granivore	LC		✓
Bronze Mannikin	*Spermestes cucullata*	Estrildidae	NF	Granivore	LC		✓
Brown-crowned Tchagra	*Tchagra australis*	Malaconotidae	NF	Insectivore	LC		✓
Common Fiscal	*Lanius collaris*	Laniidae	NF	Insectivore	LC		✓
Common Stonechat	*Saxicola torquatus*	Muscicapidae	NF	Insectivore	LC		✓
Eurasian Nightjar	*Caprimulgus europaeus*	Caprimulgidae	NF	Insectivore	LC		✓
Golden-breasted Bunting	*Emberiza flaviventris*	Emberizidae	NF	Granivore	LC		✓
Grey-headed Sparrow	*Passer griseus*	Passeridae	NF	Granivore	LC		✓
Holub's Golden Weaver	*Ploceus xanthops*	Ploceidae	NF	Granivore	LC		✓
Jackobin's Cuckoo	*Oxylophus jacobinus*	Cuculidae	NF	Insectivore	LC		✓
Plain martin	*Riparia paludicola ducis*	Hirundinidae	NF	Insectivore	LC		✓
Red-headed Weaver	*Anaplectes rubriceps*	Ploceidae	NF	Insectivore	LC		✓
Singing Cisticola	*Cisticola cantans*	Cisticolidae	NF	Insectivore	LC		✓
Speckled Mousebird	*Colius striatus*	Coliidae	NF	Frugivore	LC		✓
Yellow Wagtail	*Motacilla flava*	Motacillidae	NF	Insectivore	LC		✓

Table 6.2: Individual bird species that were unique to disturbed forest type. FD= forest dependency, FS = forest specialist, FG = forest generalist, FV = forest visitor, NF = non-forest, DG = dietary guild, ATHB = Afrotropical highlands biome species, IUCN Cat. = International Union for Conservation of Nature red list categories, and Dist. = disturbed forest.

Species	Scientific name	Family	FD	DG	IUCN Cat	ATHB	Dist.
Orange Ground Thrush	*Zoothera gurneyi*	Turdidae	FS	Insectivore	LC	✓	✓
Purple-throated Cuckoo-shrike	*Campephaga quiscalina martini*	Campephagidae	FS	Insectivore	LC		✓
Slender-billed Greenbul	*Andropadus gracilirostris*	Pycnonotidae	FS	Frugivore	LC		✓
White-browed Crombec	*Sylvietta leucophrys*	Macrosphenidae	FS	Insectivore	LC	✓	✓
African Goshawk	*Accipiter tachiro*	Accipitridae	FG	Carnivore	LC		✓
Ayres's Hawk Eagle	*Hieraaetus ayresii*	Accipitridae	FG	Carnivore	LC		✓
Oriole-Finch	*Linurgus olivaceus*	Fringillidae	FG	Granivore	LC	✓	✓
Red-headed Bluebill	*Spermophaga ruficapilla*	Estrildidae	FG	Granivore	LC		✓
Black Cuckoo-shrike	*Campephaga flava*	Campephagidae	FV	Insectivore	LC		✓
Greater Honeyguide	*Indicator indicator*	Indicatoridae	FV	Insectivore	LC		✓
Scaly-throated Honeyguide	*Indicator variegatus*	Indicatoridae	FV	Insectivore	LC		✓
African Black Duck	*Anas sparsa*	Anatidae	NF	Omnivore	LC		✓
African Fish Eagle	*Haliaeetus vocifer*	Accipitridae	NF	Carnivore	LC		✓
Black-headed Weaver	*Ploceus melanocephalus*	Ploceidae	NF	Granivore	LC		✓
Blue-headed Coucal	*Centropus monachus*	Cuculidae	NF	Carnivore	LC		✓
Giant Kingfisher	*Megaceryle maxima*	Alcedinidae	NF	Carnivore	LC		✓

Table 6.3: Individual bird species that were unique to undisturbed forest type. FD= forest dependency, FS = forest specialist, FG = forest generalist, FV = forest visitor, DG = dietary guild, ATHB = Afrotropical highlands biome species, IUCN Cat. = International Union for Conservation of Nature red list categories, and Undist. = undisturbed forest.

Species	Scientific name	Family	FD	DG	IUCN Cat	ATHB	Undist.
Mountain Buzzard	*Buteo oreophilus*	Accipitridae	FS	Carnivore	NT	✓	✓
Mountain Yellow Warbler	*Chloropeta similis*	Acrocephalidae	FG	Insectivore	LC	✓	✓
Slender-billed Starling	*Onychognathus tenuirostris theresae*	Sturnidae	FG	Omnivore	LC	✓	✓
Black Saw-wing	*Psalidoprocne holomelas*	Hirundinidae	FV	Insectivore	LC		✓

On forest dependency group, both undisturbed and disturbed forest type has more unique true forest bird species (FS and FG) and less non-true forest birds (FV and NF) (Figure 6.1 b). But disturbed forest unexpectedly has more unique FS and FG species and expectedly has more FV and NF species than undisturbed forest. As expected, eucalyptus plantation has no unique FS species and few FG species but has most unique FV and NF species to it (Figure 6.1 b). Among dietary guilds, as compared to other guilds, insectivore species were with the highest number of unique species in all the forest types (Figure 6.1 c). Eucalyptus plantation holds high number of insectivores, granivores, omnivores and nectarivores unique to it relative to other forest types. Disturbed forest on the other hand holds high number of carnivore species unique to it, while having similar number of omnivores with undisturbed forest type.

Slightly more than a third of the 40 recorded species of Afrotropical highlands biome restricted species were shared across all forest types, with a third of them shared between undisturbed and disturbed forest types (Figure 6.1 d). But very few of these species are shared between disturbed and eucalyptus plantation and undisturbed and eucalyptus plantation (Figure 6.1 d). Eucalyptus plantation had high unique number of species of ATHB group compared to unique species in undisturbed and disturbed forest types (Figure 6.1 d).

6.3.2 Variation of bird community composition across forest types

To show the variation of overall birds, forest dependency, dietary guilds and Afrotropical highlands biome species community composition across forest types, analysis of similarities (ANOSIM) was carried out. The variations were also illustrated using NMDS ordinations. Table 6.4 below shows the degree of similarity among forest types (Global *R*) and significance of variation across forest types of all bird species groups.

Table 6.4: ANOSIM of bird species groups across forest types. Global R = degree of similarity among forest types, the closer the value is to 1, the more dissimilar are the species composition. P -values shows level of significant values from 9999 permutations in PAST Version 4.02.

Bird species groups	Global *R*	P-value
Overall birds	0.54	0.0001
Forest dependency		
Forest specialists	0.43	0.0001
Forest generalists	0.29	0.0001
Forest visitors	0.26	0.0001
Non-forest birds	0.40	0.001
Dietary guilds		
Carnivores	0.009	0.4693
frugivores	0.38	0.0001
granivores	0.13	0.0001
insectivore	0.42	0.0001
Nectarivores	0.06	0.0001
omnivore	0.24	0.0001
ATHB species		
ATHB species	0.37	0.0001

a) Overall bird species and forest dependency NMDS variations

All bird species groups except for carnivore bird species differed significantly across forest types with various degree of similarity across forest types (Table 6.4). The extent of the difference in species composition across these forest types is illustrated in the NMDS ordinations in Figures 6.2-6.4 except for carnivore species composition which showed very low non-significant dissimilarity across forest types (Table 6.4). Therefore, there were no further similarity percentage analysis for carnivore dietary group between paired forest types and no comparison was made across forest types.

For overall bird species, the NMDS illustration of species composition show some overlap in overall bird species composition between undisturbed and disturbed forest types, with separating species composition in eucalyptus plantation forest type (Figure 6.2 a). Species composition is more variable within undisturbed and eucalyptus forests (not clustered) relative to within the disturbed forest.

Among forest dependency, NMDS ordination of forest specialists, shows a near overlap and almost clustering of species composition of forest specialists in undisturbed and disturbed forest types (Figure 6.2 b). There is some separation of species composition in eucalyptus plantation relative to the other forest types, with non-clustering of individual species composition. NMDS illustrate an overlapping forest generalist's species composition in undisturbed and disturbed forest, with almost separating generalist's species composition in eucalyptus plantation forest. There is close clustering of species composition of forest generalists at converging point of all forest types (Figure 6.2 c). For forest visitors, NMDS shows more overlap between the two natural forests (disturbed and undisturbed) and separation between natural forests and eucalyptus plantation. There is no clustering of forest visitors' species composition in undisturbed forest, a little clustering in disturbed forest and much clustering in eucalyptus plantation (Figure 6.2 d). The high levels of clustering (i.e., closeness of points) of forest visitors' species composition within the plantation illustrate smaller dissimilarity of it than in the undisturbed and disturbed forest types. NMDS ordination in figure 6.2 e shows non-forest species composition with more overlap between disturbed and plantation forest types. There is however no clustering of species composition across forest types, showing variability.

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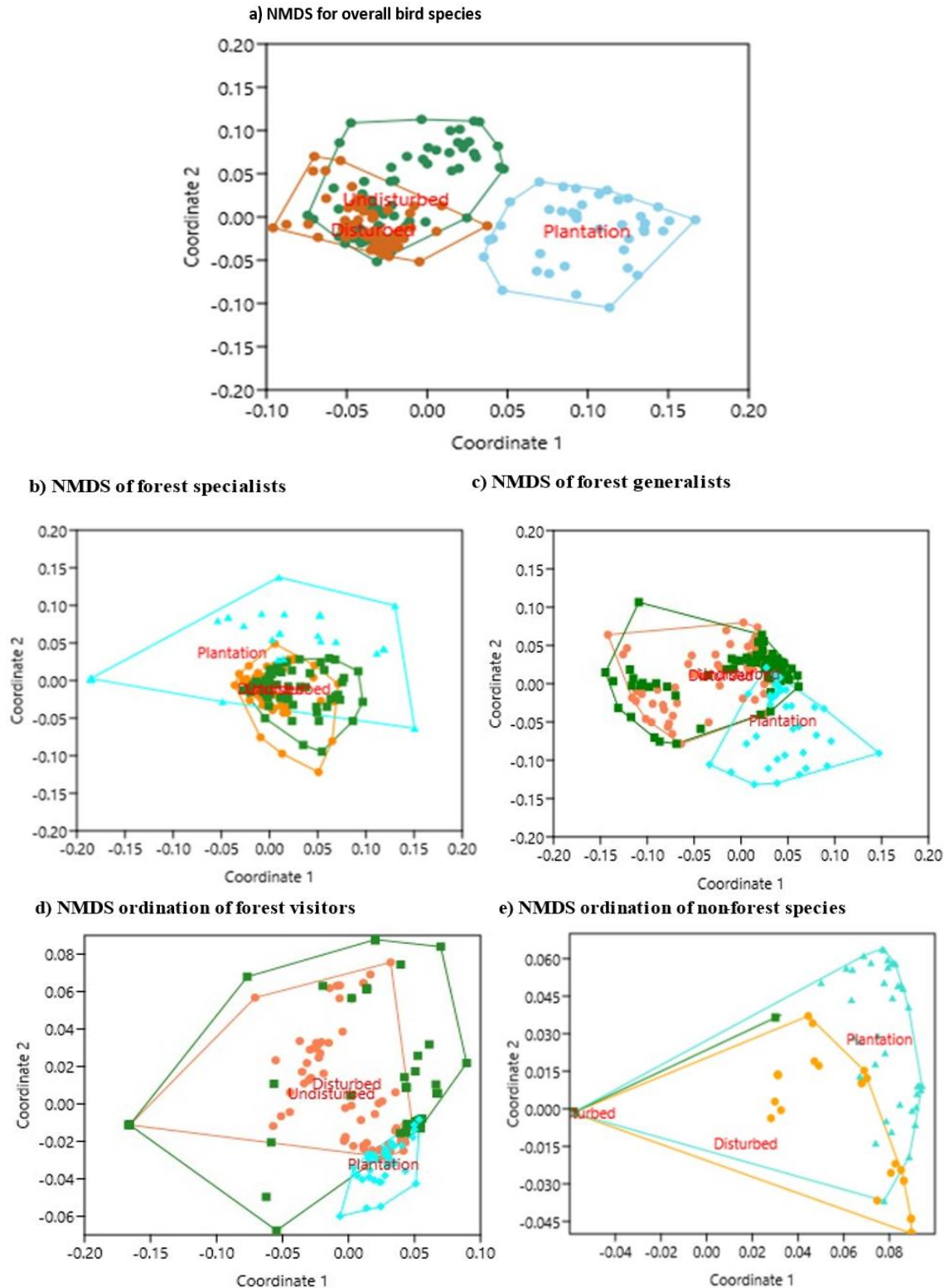
Figure 6.2: NMDS ordination for overall birds and forest dependent bird species composition across forest types (undisturbed (green), disturbed (coral) and plantation (aqua)) for (a) overall bird species, (b) forest specialists, (c) forest generalists, (d) forest visitors, (e) non-forest birds. The borderline delineates bird species composition for each forest type.

b) Dietary guilds' species composition variation in NMDS

NMDS ordination for frugivores in figure 6.3 (a), showed a complete overlap between disturbed and undisturbed forest types but although there are some overlaps with eucalyptus plantations, eucalyptus plantation mapped separately and with more variable composition than in natural forest (undisturbed and disturbed).

NMDS ordination of granivores species composition illustrate some overlap across all forest types, and more variability, but there is separation of species composition of undisturbed/disturbed forests from that of eucalyptus plantation (Figure 6.3 b). For insectivores, there is more overlaps and almost clustering of insectivore composition between undisturbed and disturbed forest types (Figure 6.3 c). However, there is complete separation of eucalyptus plantation composition from that of undisturbed, while there is some overlap with that of disturbed forest type. There is no clustering of insectivore species in the eucalyptus plantation. NMDS illustrates greater overall overlap and clustering of nectarivore species distribution points across forest types (Figure 6.3 d). It illustrates a closeness of omnivore species composition in eucalyptus plantation forest, separating from that of undisturbed and disturbed forest types in opposite direction from that of plantation forest (Figure 6.4 e).

a) NMDS ordination of frugivore species composition

b) NMDS ordination of granivore species composition

c) NMDS ordination of insectivore species composition

d) NMDS ordination of nectarivore species composition

e) NMDS ordination of omnivores species composition
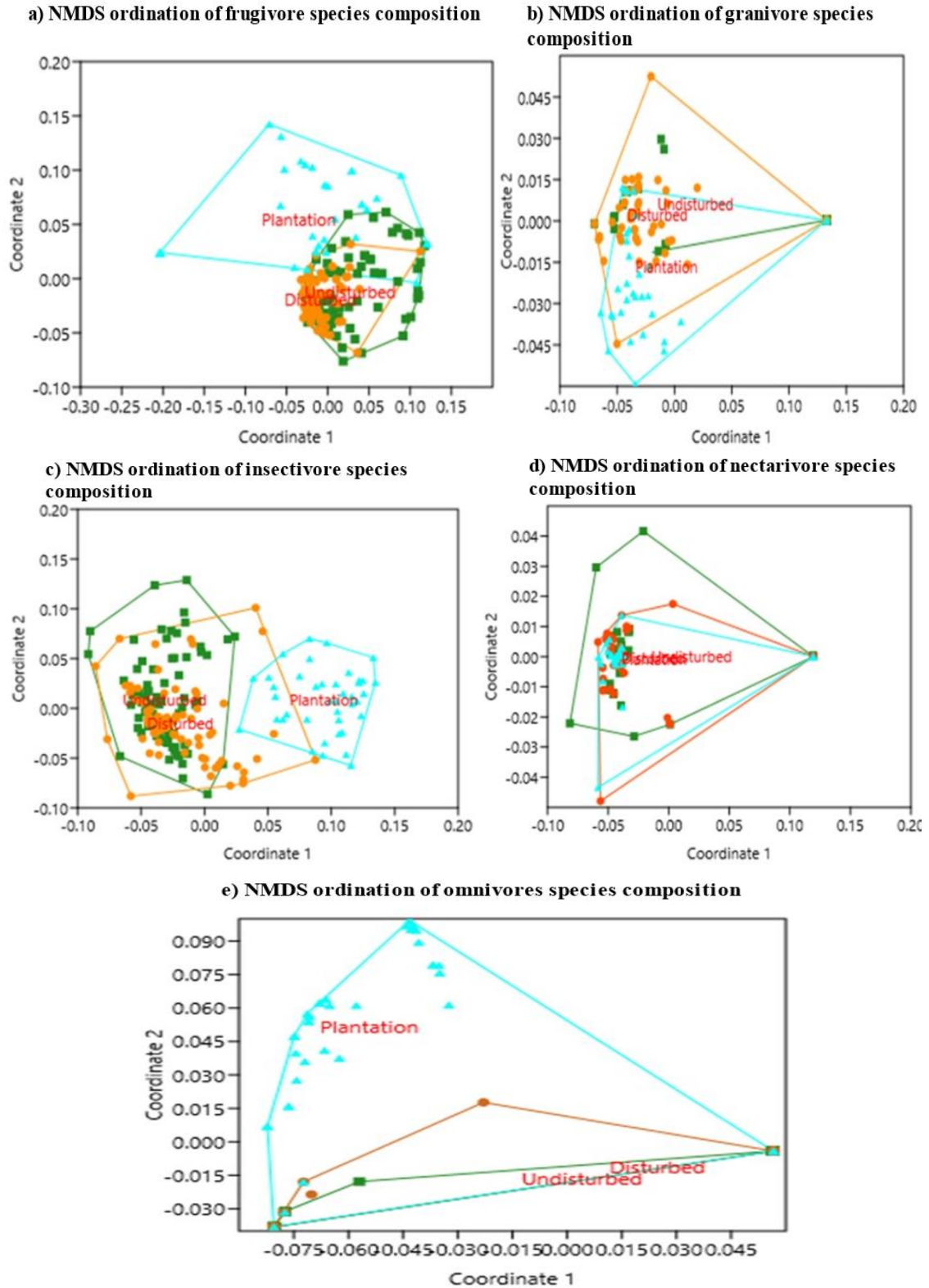
Figure 6.3: NMDS ordination of dietary guilds bird species composition across forest types (undisturbed (green), disturbed (coral) and plantation (aqua)) for (a) frugivores, (b) granivores, (c) insectivores, (d) nectarivores, and (e) omnivores. The borderline delineates bird species composition for each forest type.

c) NMDS of Afrotropical highlands biome species composition variation across forest types

NMDS illustration (Figure 6.4) shows more overlap between undisturbed and disturbed forests, more clustering within undisturbed, and some variability in disturbed forest types. Undisturbed and disturbed forests separate with eucalyptus plantation, with plantation having greater variability of ATHB species compositions.
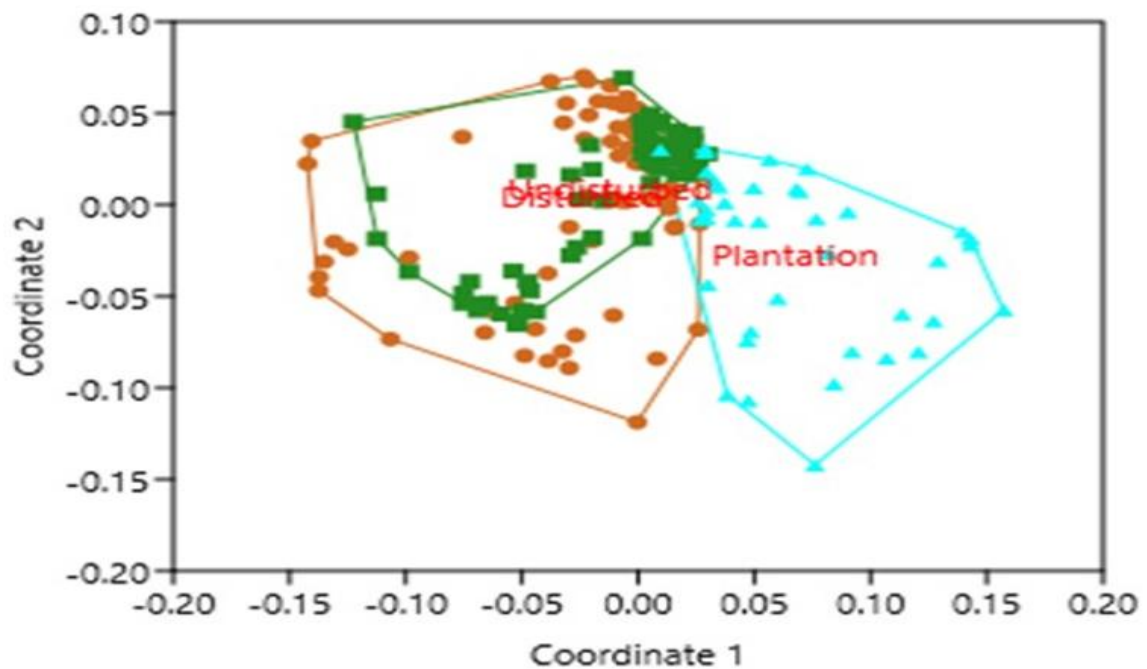
Figure 6.4: NMDS ordination of Afrotropical highlands biome species composition across undisturbed (green), disturbed (red) and plantation (aqua).

6.3.3 Individual bird species characteristics to forest types

a) Overall species characteristic to forest types

Of the overall bird species recorded, species that were characteristic to undisturbed forest (i.e., species with high mean abundance within undisturbed forest in table 6.5) based on SIMPER analysis were mainly forest specialists consisting of insectivores and frugivores. This included 6 forest specialists (3 insectivores and 3 frugivores), one forest generalist (insectivore) and one forest visitor (insectivore) (Table 6.5). Species that were

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characteristic to disturbed forest type also consisted mainly of forest specialists and generalists, but majority in the disturbed forest are frugivore species. These included 5 forest specialists (2 insectivores and 3 frugivores), 4 forest generalists (3 frugivores and one granivore) and one forest visitor (insectivore) (Table 6.5). Bird species characteristic to the eucalyptus plantation forest included both forest generalists, forest visitors and non-forest birds, all a mix of insectivores, omnivores, frugivores and granivores. For example, there are five forest generalists e.g., *Merops oreobates (*insectivore), *Melaenornis fischeri* (insectivore), (*Muscicapa adusta* (insectivore), C*isticola hunteri* (insectivore), and *Cinnyris mediocris* (nectarivore), all insectivores except *Cinnyris mediocris* (Table 6.5). For forest visitors characteristics to eucalyptus plantation included four forest visitors: *Ploceus baglafecht* (omnivore), *Pycnonotus barbatus* (omnivore), *Lonchura bicolor* (granivore), and *Laniarius major* (insectivore), while two non-forest birds included *Cisticola cantans* (insectivore), and *Colius striatus* (frugivore) (Table 6.5).

Table 6.5: Similarity percentage analysis (SIMPER) of bird abundance between undisturbed and disturbed, undisturbed and eucalyptus plantation and disturbed and eucalyptus plantation forest types in Mount Kenya forest reserve. It shows the average dissimilarity of each species between the forests and percentage contribution that each species is making to the dissimilarity between bird species composition of the forest types. It also shows the species that contribute most (i.e., species that are characteristic of each forest type) to the differences between the forest types based on high mean abundance at each forest. Overall dissimilarity between undisturbed and disturbed forests = 65.2 %, dissimilarity between undisturbed and plantation forests = 83 %, between disturbed and plantation forests = 82 %. DG means Dietary guild: INS = insectivore, FRU = frugivore, GRA = granivore, OMN = omnivore, NEC = nectarivore. FD mean Forest dependency: FS = forest specialist, FG = forest generalist, FV = forest visitors, NF = non-forest. Only species contributing to 50% of the differences in species composition between the pair of forest types shown in the table.

Scientific name	DG	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in Undisturbed	Mean abundance in Disturbed
Zosterops eurycricotus	INS	FG	5.82	8.92	8.921	4.39	4.05
Phyllastrephus cabanisi	INS	FS	2.83	4.34	13.26	1.60	2.27
Bycanistes brevis	FRU	FG	2.41	3.69	16.95	0.57	1.72
Phoeniculus bollei	INS	FS	2.12	3.25	20.20	1.21	0.95
Onychognathus walleri	FRU	FS	1.77	2.73	22.92	0.59	0.97
Eurillas latirostris	FRU	FG	1.71	2.62	25.54	1.69	2.35
Apalis jacksoni	INS	FS	1.59	2.44	27.98	0.93	0.83
Laniarius major	INS	FV	1.57	2.41	30.38	1.01	0.89
Tauraco hartlaubi	FRU	FS	1.55	2.39	32.77	1.99	1.36
Arizelocichla nigriceps	FRU	FS	1.48	2.27	35.04	1.11	0.72
Pogoniulus bilineatus	FRU	FG	1.48	2.27	37.31	0.75	0.99
Oriolus percivali	FRU	FS	1.43	2.20	39.51	1.04	1.45
Elminia albonotata	INS	FS	1.38	2.11	41.62	0.97	0.29
Terpsiphone viridis	INS	FV	1.35	2.07	43.69	0.48	1.17
Turtur tympanistria	GRA	FG	1.33	2.04	45.73	0.79	1.08
Columba arquatrix	FRU	FS	1.29	1.98	47.71	0.72	0.55
Pseudalethe poliocephala	INS	FS	1.25	1.92	49.63	0.63	0.72
Poeoptera kenricki	FRU	FS	1.23	1.88	51.51	0.47	0.71

Scientific name	DG	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in Undisturbed	Mean abundance in Plantation
Zosterops eurycricotus	INS	FG	5.36	6.43	6.43	4.39	2.20
Tauraco hartlaubi	FRU	FS	2.50	3.00	9.43	1.99	0.45
Cisticola cantans	INS	NF	2.48	2.98	12.41	0.00	1.63
Phyllastrephus cabanisi	INS	FS	2.46	2.95	15.36	1.60	0.13

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Scientific name	DG	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in Disturbed	Mean abundance in Plantation
Ploceus baglafecht	OMN	FV	2.42	2.90	18.26	0.00	1.65
Eurillas latirostris	FRU	FG	2.06	2.47	20.73	1.69	0.65
Merops oreobates	INS	FG	2.02	2.43	23.16	0.07	1.38
Pycnonotus barbatus	OMN	FV	1.96	2.35	25.51	0.00	1.35
Melaenornis fischeri	INS	FG	1.93	2.32	27.83	0.19	1.33
Lonchura bicolor	GRA	FV	1.87	2.24	30.07	0.00	1.43
Laniarius major	INS	FV	1.86	2.23	32.31	1.01	2.05
Phoeniculus bollei	INS	FS	1.77	2.12	34.43	1.21	0.00
Muscicapa adusta	INS	FG	1.64	1.97	36.40	0.51	1.38
Cisticola hunteri	INS	FG	1.61	1.94	38.33	0.52	0.90
Oriolus percivali	FRU	FS	1.61	1.93	40.26	1.04	0.45
Arizelocichla nigriceps	FRU	FS	1.56	1.87	42.13	1.11	0.18
Colius striatus	FRU	NF	1.52	1.83	43.95	0.00	1.10
Elminia albonotata	INS	FS	1.49	1.79	45.75	0.97	0.00
Apalis jacksoni	INS	FS	1.38	1.65	47.40	0.93	0.00
Cinnyris mediocris	NEC	FG	1.35	1.62	49.02	0.39	0.88
Apalis cinerea	INS	FS	1.33	1.60	50.61	0.91	0.00

Scientific name	DG	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in Disturbed	Mean abundance in Plantation
Zosterops eurycricotus	INS	FG	5.05	6.14	6.143	4.05	2.20
Phyllastrephus cabanisi	INS	FS	3.19	3.89	10.03	2.27	0.13
Eurillas latirostris	FRU	FG	2.57	3.13	13.16	2.35	0.65
Bycanistes brevis	FRU	FG	2.44	2.97	16.13	1.72	0.05
Cisticola cantans	INS	NF	2.39	2.91	19.04	0.00	1.63
Ploceus baglafecht	OMN	FV	2.33	2.84	21.88	0.00	1.65
Laniarius major	INS	FV	2.05	2.495	24.37	0.89	2.05
Merops oreobates	INS	FG	1.93	2.354	26.73	0.29	1.38
Pycnonotus barbatus	OMN	FV	1.89	2.299	29.02	0.00	1.35
Oriolus percivali	FRU	FS	1.89	2.296	31.32	1.45	0.45
Melaenornis fischeri	INS	FG	1.86	2.267	33.59	0.09	1.33
Lonchura bicolor	GRA	FV	1.81	2.202	35.79	0.00	1.43
Tauraco hartlaubi	FRU	FS	1.73	2.106	37.90	1.36	0.45
Muscicapa adusta	INS	FG	1.56	1.9	39.80	0.49	1.38
Colius striatus	FRU	NF	1.47	1.788	41.58	0.00	1.10
Pogoniulus bilineatus	FRU	FG	1.38	1.68	43.26	0.99	0.13
Turtur tympanistria	GRA	FG	1.38	1.674	44.94	1.08	0.35
Phoeniculus bollei	INS	FS	1.33	1.62	46.56	0.95	0.00
Cinnyris mediocris	NEC	FG	1.30	1.581	48.14	0.33	0.88
Onychognathus walleri	FRU	FS	1.28	1.559	49.70	0.97	0.00
Cisticola hunteri	INS	FG	1.27	1.544	51.24	0.00	0.90

b) Forest dependent bird species categories characteristics to forest types.

Out of eight species that explain 50 % of the forest specialist's species composition between undisturbed and disturbed forest types, five species were characteristic to undisturbed forest type that included three insectivores; *Phoeniculus bollei*, *Apalis jacksoni*, and *Elminia albonotata* and two frugivores that include *Tauraco hartlaubi* and *Arizelocichla nigriceps* (Table 6.6). Disturbed forest on the other hand was characterised by three forest specialist's species, one insectivore *Phyllastrephus cabanisi* and two frugivores, *Onychognathus walleri* and *Oriolus percivali* (Table 6.6).

Of the six species (3 insectivores and 3 frugivores) that explain the 50 % of the forest specialist birds composition between disturbed and eucalyptus plantation, none of them was characteristic to the eucalyptus plantation (Table 6.6).

Table 6.6: Similarity percentage analysis (SIMPER) of bird abundance of forest specialists between undisturbed and disturbed, undisturbed and eucalyptus plantation and disturbed and eucalyptus plantation forest types in Mount Kenya forest reserve. It shows average dissimilarity per species and the percentage contribution that each species makes to the dissimilarity between specialists' bird species composition of the paired forest types. It also shows forest specialist species contributing most (i.e., species that are characteristic of each forest type) to the differences between the forest types based on high mean abundance of species per forest type. DG mean Dietary guilds: INS = insectivore, FRU = frugivore. Only species contributing to 50 % of the differences in species composition between the two forests are shown in the table. Overall average dissimilarity between undisturbed and disturbed = 64 %, between undisturbed and plantation = 92 %, between disturbed and plantation = 90 %.

Scientific name	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in Undisturbed	Mean abundance in Disturbed
Phyllastrephus cabanisi	INS	6.55	10.23	10.23	1.60	2.27
Phoeniculus bollei	INS	4.79	7.47	17.69	1.21	0.95
Onychognathus walleri	FRU	3.92	6.12	23.81	0.59	0.97
Apalis jacksoni	INS	3.65	5.69	29.50	0.93	0.83
Tauraco hartlaubi	FRU	3.64	5.69	35.19	1.99	1.36
Arizelocichla nigriceps	FRU	3.45	5.38	40.56	1.11	0.72
Oriolus percivali	FRU	3.37	5.28	45.82	1.04	1.45
Elminia albonotata	INS	3.18	4.96	50.78	0.97	0.29

Scientific name	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in Undisturbed	Mean abundance in Plantation
Tauraco hartlaubi	FRU	9.86	10.72	10.72	1.99	0.45
Phyllastrephus cabanisi	INS	8.74	9.50	20.22	1.60	0.13
Phoeniculus bollei	INS	6.52	7.09	27.31	1.21	0.00
Oriolus percivali	FRU	5.91	6.43	33.74	1.04	0.45
Arizelocichla nigriceps	FRU	5.91	6.42	40.16	1.11	0.18
Elminia albonotata	INS	5.71	6.21	46.36	0.97	0.00
Apalis jacksoni	INS	5.00	5.44	51.80	0.93	0.00

Scientific name	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in disturbed	Mean abundance in plantation
Phyllastrephus cabanisi	INS	14.45	15.97	15.97	2.27	0.13
Oriolus percivali	FRU	9.02	9.97	25.93	1.45	0.45
Tauraco hartlaubi	FRU	8.55	9.45	35.38	1.36	0.45
Phoeniculus bollei	INS	5.44	6.02	41.40	0.95	0.00
Onychognathus walleri	FRU	5.17	5.71	47.11	0.97	0.00
Apalis jacksoni	INS	5.09	5.63	52.74	0.83	0.00

Seven species explains 50 % of the generalist bird's composition difference between undisturbed and disturbed forest types, but two insectivore species, Montane White-eye (*Zosterops poliogaster*) and White-starred Robin (*Pogonocichla stellata*), were characteristic to undisturbed forest type. Five species, that include three frugivores: Silvery-cheeked Hornbill (*Bycanistes brevis*), Yellow-whiskered Greenbul (*Eurillas latirostris*) and Yellow-rumped Tinkerbird (*Pogoniulus bilineatus*), and two granivores: Tambourine Dove (*Turtur tympanistria*) and Crested Guineafowl (*Guttera pucherani*) were generalists species characteristics to disturbed forest (Table 6.7).

Forest generalist's species that are characteristic to eucalyptus plantation are mainly insectivores (i.e., 4 insectivores and one nectarivore). The insectivores are: White-eyed Slaty Flycatcher *(Melaenornis fischeri)*, Cinnamon-chested Bee-eater (*Merops oreobates*), African Dusky Flycatcher (*Muscicapa adusta*) and Hunter's Cisticola (*Cisticola hunter)*, while one nectarivore is Eastern Double-collared Sunbird (*Cinnyris mediocris*) (Table 6.7).

Table 6.7: Similarity percentage analysis (SIMPER) of forest generalists bird abundance between undisturbed and disturbed, undisturbed and eucalyptus plantation and disturbed and eucalyptus plantation forest types in Mount Kenya forest reserve. It shows the percentage contribution of only forest generalists species contributing to 50 % of the difference in species composition between undisturbed and disturbed, undisturbed and eucalyptus plantation and disturbed and eucalyptus plantation forest type. It also shows contribution that each species is making to the dissimilarity between forest generalists bird species composition of the paired forest types and identify species that contribute most (i.e., species that are characteristic to each forest type) to the differences between the forest types. DG mean Dietary guilds: INS = insectivore, FRU = frugivore, GRA = granivore. Overall average dissimilarity between undisturbed and disturbed forests = 63 %, between undisturbed and plantation forests = 76 % and that between disturbed and plantation forests = 77 %.

Scientific name	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in disturbed
Zosterops eurycricotus	INS	13.10	20.73	20.73	4.39	4.05
Bycanistes brevis	FRU	5.48	8.67	29.40	0.57	1.72
Eurillas latirostris	FRU	3.90	6.18	35.57	1.69	2.35
Pogoniulus bilineatus	FRU	3.47	5.48	41.06	0.75	0.99
Turtur tympanistria	GRA	3.09	4.90	45.95	0.79	1.08
Guttera pucherani	GRA	2.24	3.54	49.49	0.19	0.56
Pogonocichla stellata	INS	2.16	3.41	52.90	0.28	0.59

Scientific name	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in plantation
Zosterops eurycricotus	INS	12.52	16.56	16.56	4.39	2.20
Eurillas latirostris	FRU	5.13	6.78	23.34	1.69	0.65
Melaenornis fischeri	INS	4.84	6.39	29.73	0.19	1.33
Merops oreobates	INS	4.82	6.37	36.10	0.07	1.38
Muscicapa adusta	INS	3.98	5.27	41.37	0.51	1.38
Cisticola hunteri	INS	3.78	5.00	46.38	0.52	0.90
Cinnyris mediocris	NEC	3.22	4.26	50.63	0.39	0.88

Scientific name	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in disturbed	Mean abundance in plantation
Zosterops eurycricotus	INS	11.92	15.52	15.52	4.05	2.20
Eurillas latirostris	FRU	6.233	8.113	23.63	2.35	0.65
Bycanistes brevis	FRU	5.746	7.479	31.11	1.72	0.05
Melaenornis fischeri	INS	4.612	6.003	37.11	0.09	1.33
Merops oreobates	INS	4.594	5.98	43.09	0.29	1.38
Muscicapa adusta	INS	3.702	4.818	47.91	0.49	1.38
Pogoniulus bilineatus	FRU	3.367	4.382	52.29	0.99	0.13

The pairwise comparison of forest types on composition of forest visitor's species showed that 50 % of the difference in species composition of forest visitors between undisturbed and disturbed forest is driven by only 3 species (2 insectivores and one frugivore). Of these, the only insectivore species that was characteristic to undisturbed forest is Tropical Boubou (*Laniarius major*). The forest visitor's characteristic to disturbed forest were African Paradise Flycatcher (*Terpsiphone viridis*) insectivore and Crowned Hornbill (*Tockus alboterminatus*) frugivore (Table 6.8). Forest visitor's species characteristic to eucalyptus plantation forest included a mix of dietary guilds that include Baglafecht Weaver (*Ploceus baglafecht*) omnivore, Tropical Boubou (*Laniarius major*) insectivore, Common Bulbul (*Pycnonotus barbatus*) omnivore, Black-and-white Mannikin (*Lonchura bicolor*) granivore (Table 6.8).

For non-forest birds, none were characteristic to undisturbed forest type. But Chin-Spot Batis (*Batis molitor*) insectivore, was the only species that was characteristics to disturbed forest (Table 6.9). However, four non-forest bird species were characteristic to eucalyptus plantation forest namely Singing Cisticola (*Cisticola cantans*) insectivores, Common Fiscal (*Lanius collaris*) insectivore, and Chin-spot Batis (*Batis molitor*) insectivore and Speckled Mousebird (*Colius striatus*) frugivore (Table 6.9).

Table 6.8: Similarity percentage analysis (SIMPER) of forest visitors bird abundance between undisturbed and disturbed, undisturbed/disturbed and plantation forest types in Mount Kenya forest reserve. It shows the average similarity and percentage contribution of forest visitors species contributing to 50 % of the difference in species composition between undisturbed and disturbed, and undisturbed/disturbed and plantation forest type. It also shows contribution that each species is making to the dissimilarity between the two forest types and identify species that contribute most (i.e., species that are characteristic to each forest type) to the differences between undisturbed and disturbed, and the undisturbed/disturbed and plantation forest types based on high mean abundance of each species in each forest type. DG mean Dietary guilds: INS = insectivore, FRU = frugivore, OMN = omnivore, GRA = granivore. Overall average dissimilarity between undisturbed and disturbed = 79 %, undisturbed and plantation forests = 81 % and between disturbed and plantation forest = 78 %.

Scientific name	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in disturbed	Mean abundance in undisturbed
Terpsiphone viridis	INS	16.84	21.34	21.34	1.17	0.48
Laniarius major	INS	16.66	21.12	42.46	0.89	1.01
Lophoceros alboterminatus	FRU	9.42	11.93	54.40	0.75	0.11

Scientific name	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in plantation
Ploceus baglafecht	OMN	12.3	15.17	15.17	0.00	1.65
Laniarius major	INS	10.59	13.06	28.23	1.01	2.05
Pycnonotus barbatus	OMN	9.598	11.84	40.07	0.00	1.35
Terpsiphone viridis	INS	6.91	8.523	48.59	0.48	1.00
Lonchura bicolor	GRA	6.706	8.271	56.86	0.00	1.43

Scientific name	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in disturbed	Mean abundance in plantation
Ploceus baglafecht	OMN	10.19	13.05	13.05	0.00	1.65
Laniarius major	INS	9.74	12.47	25.52	0.89	2.05
Pycnonotus barbatus	OMN	8.03	10.28	35.81	0.00	1.35
Lonchura bicolor	GRA	5.96	7.63	43.43	0.00	1.43
Terpsiphone viridis	INS	4.40	5.64	49.07	1.17	1.00
Lophoceros alboterminatus	FRU	4.36	5.58	54.65	0.75	0.05

Table 6.9: Similarity percentage analysis (SIMPER) of non-forest bird abundance between undisturbed and disturbed, undisturbed and plantation and disturbed and plantation forest types in Mount Kenya forest reserve. It shows the average dissimilarity and percentage contribution of non-forest bird species contributing to 50 % of the difference in species composition between undisturbed and disturbed, undisturbed and planation, and disturbed and plantation forest types. It also shows contribution of one species that is making to the dissimilarity between the paired forest types and identify where the species contribute most (i.e., where it is more associated with in terms of forest type) to the differences between the paired forest types. DG mean Dietary guilds: INS = insectivore.

Scientific names	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance Undisturbed	Mean abundance Disturbed
Batis molitor	INS	30.96	53.47	53.47	0	0.4

Scientific names	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in plantation
Cisticola cantans	INS	25.96	26.62	26.62	0.00	1.63
Lanius collaris	INS	13.11	13.44	40.07	0.00	0.75
Colius striatus	INS	11.55	11.85	51.91	0.00	1.10

Scientific names	DG	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean Disturbed	Mean Plantation
Cisticola cantans	INS	22.34	23.72	23.72	0.00	1.63
Lanius collaris	INS	10.89	11.57	35.28	0.00	0.75
Colius striatus	FRU	10.42	11.06	46.35	0.00	1.10
Batis molitor	INS	9.88	10.49	56.84	0.40	0.55

c) Dietary guilds' species composition characteristics to forest types.

Frugivores species that are characteristics to undisturbed forest are all forest specialists that included Harlaub's Turaco (*Turaco Hartlaubi*), and Mountain Greenbul (*Andropadus nigriceps*). The frugivore species characteristic to disturbed forest type are a mix of forest specialists and generalists, i.e., two forest specialists namely Waller's Starling (*Onychognathus walleri*) and Montane Oriole (*Oriolus percivali*), and two forest generalists namely Silvery-cheeked Hornbill (*Bycanistes brevis*) and Yellow-whiskered Greenbul (*Eurillas latirostris*) (Table 6.10). The only frugivore species that is characteristic to plantation forest is a non-forest Speckled Mousebird (*Colius striatus*).

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Table 6.10: Similarity percentage analysis (SIMPER) for frugivore bird species abundance between undisturbed and disturbed and between undisturbed/disturbed and plantation forest types in Mount Kenya forest reserve. It shows the average dissimilarity and the percentage contribution of frugivore bird species contributing to 50% of the difference in species composition between undisturbed and disturbed and between undisturbed/disturbed and plantation forest type. It also shows contribution that each species is making to the dissimilarity between the paired forest types and identify species that contribute most (i.e., species that are characteristic to each forest type) to the differences between the undisturbed and disturbed and between undisturbed/disturbed and plantation forest types. FD mean forest dependency: FS = forest specialist, FG = forest generalist and NF = non-forest. Overall average dissimilarity between undisturbed and disturbed forest = 57 %, undisturbed and plantation forests = 82 % and between disturbed and plantation forest = 83 %.

Scientific names	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in disturbed
Bycanistes brevis	FG	7.18	12.49	12.49	0.57	1.72
Eurillas latirostris	FG	5.64	9.82	22.30	1.69	2.35
Tauraco hartlaubi	FS	5.07	8.82	31.12	1.99	1.36
Onychognathus walleri	FS	5.01	8.72	39.84	0.58	0.97
Oriolus percivali	FS	4.76	8.29	48.13	1.04	1.45
Andropadus nigriceps	FS	4.72	8.21	56.33	1.11	0.72

Scientific names	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in plantation
Tauraco hartlaubi	FS	15.35	18.73	18.73	1.99	0.45
Eurillas latirostris	FG	11.37	13.86	32.59	1.69	0.65
Andropadus nigriceps	FS	8.88	10.83	43.43	1.11	0.18
Oriolus percivali	FS	7.93	9.67	53.09	1.04	0.45

Scientific names	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in disturbed	Mean abundance in plantation
Eurillas latirostris	FG	12.32	14.87	14.87	2.35	0.65
Bycanistes brevis	FG	10.33	12.47	27.34	1.72	0.05
Oriolus percivali	FS	9.14	11.04	38.38	1.45	0.45
Tauraco hartlaubi	FS	8.49	10.25	48.63	1.36	0.45
Colius striatus	NF	6.20	7.49	56.12	0.00	1.10

50 % of the difference in species composition of granivores birds between undisturbed and disturbed forest type was determined by two forest generalist species (Tambourine Dove (*Turtur tympanistria*), and Scaly francolin (*Pternistis squamatus*), and both are characteristic to disturbed forest type. 50 % of the difference in species composition of granivores between undisturbed and eucalyptus plantation, and between disturbed and eucalyptus plantation forest types birds were driven by four species each (i.e., Tambourine dove (*Turtur tympanistria*) forest generalist, Black-and-white Mannikin (*Lonchura bicolor*) forest visitor, Streaky Seed-eater (*Serinus striolatus*) forest visitor, and Brimstone Canary *(Serinus sulphuratus*) non-forest; and Tambourine Dove, Black-and-white Mannikin, Red-eyed Dove (*Streptopelia semitorquata*) forest visitor, and Crested Guineafowl (*Guttera pucherani*) forest generalist respectively. None of these species was characteristic to undisturbed forest but Black-and-white Mannikin, Streaky Seed-eater, and Brimstone Canary were all characteristic to eucalyptus plantation forest (Table 6.11).

50 % of the difference in species composition of insectivores between undisturbed and disturbed forest type is determined by 9 species (6 forest specialists, one generalist, and 2 forest visitors). Between undisturbed and eucalyptus plantation forests is determined by 11 species (4 forest specialists, 5 forest generalists, one forest visitor and one non-forest), while 50 % of the difference in species composition of insectivores between disturbed and plantation is determined by 10 species (3 forest specialists, 5 forest generalists, one forest visitor and one non-forest) (Table 6.12).

Table 6.11: Similarity percentage analysis (SIMPER) for granivore bird species abundance between undisturbed and disturbed and between undisturbed/disturbed and plantation forest types in Mount Kenya forest reserve. It shows the percentage contribution of granivore bird species contributing to 50 % of the difference in species composition between undisturbed and disturbed and between undisturbed/disturbed and plantation forest type. It also shows average dissimilarity and percentage contribution that each species is making to the dissimilarity between the paired forest types. It shows granivore species that contribute most (i.e., species that are characteristic to each forest type) to the differences between the undisturbed and disturbed and between undisturbed/disturbed and plantation forest types. FD mean forest dependency: FG = forest generalist, FV = forest visitors and NF = non-forest. Overall average dissimilarity between undisturbed and disturbed forest = 71 %, undisturbed and plantation forests = 83 % and between disturbed and plantation forest = 87 %.

Scientific name	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in disturbed
Turtur tympanistria	FG	31.11	43.67	43.67	0.79	1.08
Pternistis squamatus	FG	10.35	14.53	58.21	0.15	0.31

Scientific name	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in plantation
Turtur tympanistria	FG	23.37	28.02	28.02	0.787	0.35
Lonchura bicolor	FV	10.1	12.11	40.13	0	1.43
Serinus striolatus	FV	7.009	8.403	48.54	0	0.625
Serinus sulphuratus	NF	5.315	6.372	54.91	0	0.3

Scientific name	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in disturbed	Mean abundance in plantation
Turtur tympanistria	FG	19.08	21.99	21.99	1.08	0.35
Lonchura bicolor	FV	9.034	10.41	32.4	0	1.43
Streptopelia semitorquata	FV	9.017	10.39	42.8	0.48	0.375
Guttera pucherani	FG	6.446	7.43	50.23	0.56	0.1

Table 6.12: Similarity percentage analysis (SIMPER) for insectivore bird species abundance between undisturbed and disturbed and between undisturbed/disturbed and plantation forest types in Mount Kenya forest reserve. It shows the percentage contribution of insectivore bird species contributing to 50 % of the difference in species composition between undisturbed and disturbed and between undisturbed/disturbed and plantation forest type. It also shows average dissimilarity of each species as well as percentage contribution that each species is making to the overall dissimilarity between the paired forest types. It identifies species that contribute most (i.e., species that are characteristic to each forest type) to the differences between the undisturbed and disturbed and between undisturbed/disturbed and plantation forest types. FD mean forest dependency: FS = forest specialist, FG = forest generalist. FV = forest visitors and NF = non-forest. Overall average dissimilarity between undisturbed and disturbed forest = 68 %, undisturbed and plantation forests = 81 % and between disturbed and plantation forest = 78 %.

Scientific names	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in disturbed
Zosterops eurycricotus	FG	10.54	15.55	15.55	4.39	4.05
Phyllastrephus cabanisi	FS	5.24	7.74	23.29	1.60	2.27
Phoeniculus bollei	FS	3.83	5.65	28.94	1.21	0.95
Apalis jacksoni	FS	2.98	4.39	33.33	0.93	0.83
Laniarius major	FV	2.96	4.37	37.70	1.01	0.89
Elminia albonotata	FS	2.65	3.91	41.62	0.97	0.29
Terpsiphone viridis	FV	2. 50	3.69	45.30	0.48	1.17
Pseudalethe poliocephala	FS	2.36	3.49	48.79	0.63	0.72
Apalis cinerea	FS	2.21	3.26	52.06	0.91	0.44

Scientific names	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in plantation
Zosterops eurycricotus	FG	9.13	11.23	11.23	4.39	2.20
Phyllastrephus cabanisi	FS	4.34	5.34	16.56	1.60	0.12
Cisticola cantans	NF	4.29	5.27	21.84	0.00	1.63
Melaenornis fischeri	FG	3.47	4.26	26.10	0.19	1.33
Merops oreobates	FG	3.47	4.26	30.36	0.07	1.38
Laniarius major	FV	3.22	3.96	34.32	1.01	2.05
Phoeniculus bollei	FS	2.91	3.58	37.90	1.21	0.00
Muscicapa adusta	FG	2.85	3.50	41.40	0.51	1.38
Cisticola hunteri	FG	2.76	3.40	44.79	0.52	0.90
Elminia albonotata	FS	2.72	3.35	48.14	0.97	0.00
Apalis jacksoni	FS	2.45	3.01	51.15	0.93	0.00

Scientific names	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in disturbed	Mean abundance in plantation
Zosterops eurycricotus	FG	9.28	11.87	11.87	4.05	2.20

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Phyllastrephus cabanisi	FS	5.91	7.56	19.43	2.27	0.13
Cisticola cantans	NF	4.46	5.71	25.14	0.00	1.63
Laniarius major	FV	3.78	4.83	29.97	0.89	2.05
Melaenornis fischeri	FG	3.61	4.61	34.58	0.09	1.33
Merops oreobates	FG	3.58	4.57	39.15	0.29	1.38
Muscicapa adusta	FG	2.91	3.72	42.88	0.49	1.38
Phoeniculus bollei	FS	2.44	3.12	46.00	0.95	0.00
Cisticola hunteri	FG	2.37	3.03	49.03	0.00	0.90
Apalis jacksoni	FS	2.18	2.78	51.81	0.83	0.00

Majority of insectivore bird species characteristic to undisturbed forest are forest specialists that include White-headed Wood-hoopoe (*Phoeniculus bollei)*, White-tailed Crested Flycatcher (*Elminia albonotata*), Grey Apalis (*Apalis cinerea*), and Black-throated Apalis (*Apalis jacksoni*). One forest generalist, Montane White-eye (*Zosterops poliogaster*) and one forest visitor, Tropical Boubou (*Laniarius major*) are other insectivore species characteristic to undisturbed forest (Table 6.12).

 Insectivore species characteristic to disturbed forest included two forest specialist, Cabanis's Greenbul (*Phyllastrephus cabanisi*) and Brown-chested Alethe (*Pseudalethe poliocephala*), and one forest visitor: African Paradise Flycatcher (*Terpsiphone viridis*).

Insectivores' species characteristic to eucalyptus plantation forest includes 4 forest generalists: White-eyed Slaty Flycatcher (*Melaenornis fischeri*), Cinnamon-chested Bee-eater (*Merops oreobates*), African Dusky Flycatcher (*Muscicapa adusta*), and Hunter's Cisticola (*Cisticola hunteri*); and one forest visitor, Tropical Boubou (*Laniarius major*) and one non-forest species, Singing Cisticola (*Cisticola cantans*) (Table 6.12).

50 % of the difference in species composition of nectarivore birds were driven by only two forest generalist species (Eastern Double-collared Sunbird (*Cinnyris mediocris*) and Northern Double-collared Sunbird (*Cinnyris reichenowi*) in all paired combination of forest types. None of the nectarivore bird species was characteristic to undisturbed forest but only one species each characterised disturbed (Northern Double-collared Sunbird (*Cinnyris reichenowi*)) and eucalyptus plantation (Eastern Double-collared Sunbird (*Cinnyris mediocris*) (Table 6.13).

Table 6.13: Similarity percentage analysis (SIMPER) for nectarivore bird species abundance between undisturbed and disturbed and between undisturbed/disturbed and plantation forest types in Mount Kenya forest reserve. It shows the percentage contribution of nectarivore bird species contributing to 50% of the difference in species composition between undisturbed and disturbed and between undisturbed/disturbed and plantation forest type. It also shows contribution that each species is making to the dissimilarity between the paired forest types and identify species that contribute most (i.e., species that are characteristic to each forest type) to the differences between the undisturbed and disturbed and between undisturbed/disturbed and plantation forest types. FD mean forest dependency: FS = forest specialist, FG = forest generalist. FV = forest visitors and NF = non-forest. Overall average dissimilarity between undisturbed and disturbed forest = 83 %, undisturbed and plantation forests = 83 % and between disturbed and plantation forest = 85 %.

Scientific name	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in disturbed
Cinnyris reichenowi	FG	23.79	28.76	28.76	0.24	0.48
Cinnyris mediocris	FG	20.75	25.09	53.85	0.387	0.33
Scientific name	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in plantation
Cinnyris mediocris	FG	29.27	35.36	35.36	0.387	0.88
Cinnyris reichenowi	FG	13.53	16.34	51.71	0.24	0.30
Scientific name	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in disturbed	Mean abundance in plantation
Cinnyris mediocris	FG	26.08	30.80	30.80	0.33	0.88
Cinnyris reichenowi	FG	19.46	22.98	53.78	0.48	0.30

SIMPER analysis of paired forest types shows almost average dissimilarities in omnivore species composition between undisturbed and disturbed forests (i.e., 46 %) but greater dissimilarities between both undisturbed and eucalyptus plantation (88 %) and disturbed and eucalyptus plantation (91 %) forest types. But only African Black Duck (*Anas sparsa*), was a non-forest bird species that was characteristic to undisturbed forest (Table 6.14). All forest generalist omnivores (e.g., Olive Thrush (*Turdus olivaceus*) and Slender-billed Starling *(Onychognathus tenuirostris)*), and forest visitors omnivores (e.g., Cape Robin Chat (*Cossypha caffra*), Baglafecht Weaver (*Ploceus baglafecht*) and Common Bulbul (*Pycnonotus barbatus*)) were characteristic to eucalyptus plantation.

Table 6.14: Similarity percentage analysis (SIMPER) for omnivore bird species abundance between undisturbed and disturbed, and between undisturbed/disturbed and plantation forest types in Mount Kenya forest reserve. It shows the percentage contribution of omnivore bird species contributing to the difference in species composition between undisturbed and disturbed and between undisturbed/disturbed and plantation forest type. It also shows contribution that each species is making to the dissimilarity between the paired forest types and identify species that contribute most (i.e., species that are characteristic to each forest type) to the differences between the undisturbed and disturbed and between undisturbed/disturbed and plantation forest types. FD mean forest dependency: FG = forest generalist, FV = forest visitors and NF = non-forest. Overall average dissimilarity between undisturbed and disturbed forest = 46 %, undisturbed and plantation forests = 88 % and between disturbed and plantation forest = 91 %.

Scientific name	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in disturbed
Turdus olivaceus	FG	43.38	94.59	94.59	0.44	0.29
Anas sparsa	NF	1.14	2.49	97.09	0.00	0.03
Onychognathus tenuirostris	FG	0.94	2.05	99.14	0.04	0.00
Cossypha caffra	FV	0.40	0.86	100.00	0.00	0.01
Ploceus baglafecht	FV	0.00	0.00	100.00	0.00	0.00
Pycnonotus barbatus	FV	0.00	0.00	100.00	0.00	0.00

Scientific name	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in plantation
Ploceus baglafecht	FV	31.75	36.27	36.27	0.00	1.65
Pycnonotus barbatus	FV	29.15	33.30	69.57	0.00	1.35
Turdus olivaceus	FG	20.70	23.64	93.21	0.44	0.85
Cossypha caffra	FV	5.40	6.18	99.39	0.00	0.25
Onychognathus tenuirostris	FG	0.53	0.61	100.00	0.04	0.00
Anas sparsa	NF	0.00	0.00	100.00	0.00	0.00

Scientific name	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in disturbed	Mean abundance in plantation
Ploceus baglafecht	FV	32.72	36.12	36.12	0.00	1.65
Pycnonotus barbatus	FV	30.19	33.33	69.44	0.00	1.35
Turdus olivaceus	FG	21.44	23.66	93.11	0.29	0.85
Cossypha caffra	FV	5.734	6.33	99.44	0.01	0.25
Anas sparsa	NF	0.5108	0.56	100.00	0.02	0.00
Onychognathus tenuirostris	FG	0.00	0.00	100.00	0.00	0.00

d) Species of Afrotropical highlands biome that are characteristics to each forest type.

There is greater species composition dissimilarity of Afrotropical highlands biome between natural forests (undisturbed and disturbed) and eucalyptus plantation, but less dissimilarity between the two natural forests, undisturbed and disturbed forest types (Table 6.15). For example, SIMPER analysis showed overall average dissimilarity of 63 % between undisturbed and disturbed, and 81 % for both undisturbed and plantation, and between disturbed and plantation forest types. Only forest specialists and generalists were characteristics to undisturbed forest type. These are 2 forest specialists: Hartlaub's Turaco (*Tauraco hartlaubi*) frugivore and White-tailed Crested Flycatcher (*Elminia albonotata*) insectivore, and one forest generalist, Montane White-eye (*Zosterops eurycricotus*) insectivore (Table 6.15).

Species that were characteristic to disturbed forest were only forest specialists that included Waller's Starling (*Onychognathus walleri*), Montane Oriole (*Oriolus percivali*) and Kenrick's Starling (*Poeoptera kenricki*), all frugivores (Table 6.15).

Species of ATHB that were characteristic to the eucalyptus plantation forest included both forest generalists and forest visitors. These included Cinnamon-chested Bee-eater *(Merops oreobates)* insectivore and White-eyed Slaty Flycatcher (*Melaenornis fischeri*) insectivore for generalists, and one forest visitor, Baglafecht Weaver (*Ploceus baglafecht*) omnivore (Table 6.15). Only insectivores, omnivores and frugivores of ATHB are characteristics to forest types found in Mount Kenya.

Table 6.15: Similarity percentage analysis (SIMPER) for ATHB bird species abundance between undisturbed and disturbed, undisturbed and plantation, and disturbed and plantation forest types in Mount Kenya forest reserve. It shows the average dissimilarity and percentage contribution of the species, contributing to 50 % of the difference in species composition between undisturbed and disturbed, undisturbed and plantation, and disturbed and plantation forest type. It also shows contribution that each species is making to the dissimilarity between the paired forest types and identify species that contribute most (i.e., species that are characteristic to each forest type) to the differences between the undisturbed and disturbed forest types. FD mean forest dependency: FS = forest specialist, FG = forest generalist. Overall average dissimilarity between undisturbed and disturbed forest = 63 %, Overall dissimilarity between undisturbed and plantation forests = 81 % and between disturbed and plantation forest = 81 %.

Scientific name	DG	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in disturbed
Zosterops eurycricotus	INS	FG	14.20	22.58	22.58	4.39	4.05
Onychognathus walleri	FRU	FS	4.34	6.90	29.47	0.59	0.97
Tauraco hartlaubi	FRU	FS	4.00	6.37	35.84	1.99	1.36
Oriolus percivali	FRU	FS	3.65	5.80	41.64	1.04	1.45
Elminia albonotata	INS	FS	3.63	5.77	47.41	0.97	0.29
	FRU	FS	3.02	4.80	52.20	0.47	0.71

Scientific name	DG	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in undisturbed	Mean abundance in plantation
Zosterops eurycricotus	INS	FG	12.96	16.00	16.00	4.39	2.20
Tauraco hartlaubi	FRU	FS	6.32	7.80	23.80	1.99	0.45
Ploceus baglafecht	OMN	FV	6.13	7.57	31.37	0.00	1.65
Merops oreobates	INS	FG	5.31	6.56	37.93	0.07	1.38
Melaenornis fischeri	INS	FG	4.97	6.14	44.06	0.18	1.33
Oriolus percivali	FRU	FS	4.23	5.22	49.29	1.04	0.45
Elminia albonotata	INS	FS	3.92	4.84	54.13	0.97	0.00

Scientific name	DG	FD	Average dissimilarity	Percentage contribution to overall dissimilarity	Cumulative percentage of dissimilarity	Mean abundance in disturbed	Mean abundance in plantation
Zosterops eurycricotus	INS	FG	13.56	16.78	16.78	4.05	2.2
Ploceus baglafecht	OMN	FV	6.626	8.197	24.97	0	1.65
Merops oreobates	INS	FG	5.686	7.033	32.01	0.293	1.38
Melaenornis fischeri	INS	FG	5.37	6.643	38.65	0.0933	1.33
Oriolus percivali	FRU	FS	5.346	6.613	45.26	1.45	0.45
Tauraco hartlaubi	FRU	FS	4.878	6.034	51.3	1.36	0.45

6.4 Discussion

6.4.1 Shared and unique species composition across and between forest types

The occurrence of high number of shared species composition among all forest types, rather than shared between any of the paired forest types in Mount Kenya forest reserve shows that these forests together contribute positively to overall bird composition in Afromontane region. It depicts how they are likely to be commonly providing opportunities (resources and conditions) required by majority of birds in the region. That is, the resources and conditions obtained from one forest type, could also be in addition to or supplemented by the resources obtained from the other for majority of the birds. Therefore, these forests together are likely to offer more survival opportunities to many birds than when a pair or when one forest type exist. In other words, forest types are better off when they are all represented. Forest generalists in this case are likely to be the most beneficiaries, especially with the current scenario with major human impacts on forests and other ecosystems. It is not surprising therefore, to have majority of forest generalists being the most shared species across all forest types in this study (Figure 6.1 b). Similarly, the reason for insectivore species being the most shared dietary group in this study is likely because they have different foraging strategies and feeding stratum (e.g., aerial gleaners, foliage gleaners, bark foragers (e.g., Castaño-Villa et al., 2019)), hence found in a range of forest types. In addition, most of the Afrotropical highlands biome restricted species are shared among all forest types (i.e., 13 out 40 species or 32.5 %), depicting that all forest types irrespective of local disturbance of natural forests, and existing exotic plantations in the study area, all potentially contribute to the presence and probably conservation of biome restricted species in Afromontane forests. The existence of the different forest types in Afromontane landscape is creating a diversity of habitats i.e., spatial habitat heterogeneity within and between forest types (e.g., Lengyel et al., 2016), which is often invoked as a driver of species diversity at a small/local spatial scale, rather than at regional scale which is driven by species-area relationship (Böhning-Gaese, 1997). The habitat diversity is responsible for influencing species diversity and that is what is occurring in Afromontane forest of Mount Kenya. Probably, the introduced exotic plantation, though not really proving its suitability in biodiversity conservation, especially

for native forest birds, it is emerging to just to act as a non-forest habitat by attracting mostly non-forest bird species and generalists. In another perspective therefore, the eucalyptus plantation may be seen as slightly degrading habitat since its quality is not suitable for true forest dependent birds.

The shared species composition of the paired natural forest types also reveals their unmatched combined importance in species composition in Mount Kenya montane forest. For example, disturbed and undisturbed natural forest may differ in levels of local disturbance or in habitat structural characteristics as observed in Chapter 4, but their different qualities seem to enhance or emphasise the overall qualities of the natural forests. This is seen through their combined composition of bird species. For instance, undisturbed and disturbed forest both shared 33 species, yet each one uniquely holds less species, 4 and 16 species respectively (Figure 6.1 a). The majority of the 33 shared species between undisturbed and disturbed forest types are forest specialists and generalists (23 and 9 species respectively) (Figure 6.1 b), and mainly composed of insectivores and frugivores (Figure 6.1 c). In addition, both undisturbed and disturbed forest types shared most of the species' composition of highlands biome restricted species, ATHB. The shared ATHB restricted range species between undisturbed and disturbed forests, were far more than shared between either of undisturbed/disturbed and eucalyptus plantation. All these could be in response to similar habitat qualities and characteristics or that 'naturalness' that attracts these bird and bird groups to each of the forest types, but not present between a natural forest and an exotic plantation. This means the nature of forest types also have a role in species composition, and in extension conservation of species, with plantation being a less favourable choice.

The number of species of birds exclusively found in each forest type (as unique species to the forest types), possibly portrays the fidelity of those species to the forests, depending on the presence of life supporting resources and conditions that those birds require and provided by the forest type. Among natural forests (disturbed and undisturbed), the exceptionally high number of unique species to disturbed (up to 4 times the one in undisturbed forest) is likely because of additional group of bird species in disturbed forest, particularly of FV and NF than the same species in undisturbed forest. The more species

of FS and FG than in undisturbed forest could be because of more niche opportunities resulting from local disturbance and an increase in FHD recorded to characterise this forest type (Chapter 4). FV and NF are attracted to disturbed forest type by conditions resulting from disturbance making the sites more suitable for them (Bennun et al., 1996).

The unique species that was recorded in eucalyptus plantation, which are more than in any other forest type is because of presence of FV and NF (Table 6.1) that are likely attracted from surrounding non-forest habitats. These are non-true forest bird species that do not entirely depends on forest and are almost always more common in non-forest habitats (Bennun et al., 1996). They are likely visiting eucalyptus plantation because it has conditions and resources like the one found in non-forest habitats. For instance, among the most recorded dietary guilds in eucalyptus plantation include insectivores and granivores as compared to in disturbed and undisturbed forest types. They are likely more here because they have a greater range of resources or more opportunities for resources to utilise than in other forest types. These are resources that can be found more in highly disturbed or non-forest habitats like farmlands and grasslands (e.g., Ndang'ang'a et al., 2013). Therefore, eucalyptus plantation that was found to be characterised by open canopies, tallest trees, and high undergrowth vegetation (e.g., Chapter 4) could be encouraging availability of more insectivore species, probably those with diverse feeding strategies (e.g., aerial or leaf gleaning, removal from barks and trunks etc). Eucalyptus plantation could also be offering more grassy sites for granivores due to their more open canopies that allows more light penetration to forest floor that encourages grass and forbs development that bear grain seeds for birds.

Eucalyptus plantation also exclusively had 5 species belonging to ATHB, a number which is more than found in any other forest type. It was expected that the composition of these highlands biome restricted species could be more within undisturbed or disturbed forest types, owing to possible specialised adaptation of these birds to natural forests in a biome where they are restricted, and only likely influence on their composition could have been forest degradation such as disturbance. But it seems exotic plantations can also accommodate some restricted range species in Afromontane forests, although these consist mainly of FV and NF and some FG. This nevertheless reveal some conservation

value of eucalyptus plantations, and a general suitability of eucalyptus plantations to birds that are not truly forest birds (e.g., FV and NF). Ferenc et al. (2016) had noted that range-restricted bird species are unusually abundant in Afrotropical montane areas, and based on ATHB species in this study, it can also be argued that these species can evidently occupy emerging exotic vegetations such as eucalyptus plantations. This finding also supports the argument by Matuoka et al. (2020) that forest and non-forest birds shows divergent responses to forest loss in the tropics. Exotic plantations forests could be attracting otherwise non-forest birds closer to the forests ecosystem or supporting them, easing their dependence on other non-forest habitats such as farmlands. Exotic plantation therefore may partly offset the likely loss of Afromontane biodiversity from destruction of natural areas such as grasslands (e.g., Barlow et al., 2007 (b)).

In summary, it is clear from the finding of this study that overall bird species composition and those restricted to the highland's biome are shared across all forest types in a manner that they complement each other. However, natural forests, irrespective of the level of local disturbance, is particularly important in holding species composition of true forest birds (FS and FG), highland biome restricted birds, and maintenance of insectivores and frugivores bird species composition. From this, existence of exotic plantation (e.g., eucalyptus) also enhances overall assemblage of bird species composition in Afrotropical montane forests, but by just attracting otherwise non-forest birds from other non-forest habitats.

6.4.2 Bird species composition across forest types

ANOSIM showed significant differences in distribution of overall species, forest dependency, species of Afrotropical highlands biome, and all dietary groups except for carnivore birds across forest types. This implies each forest type has a role in maintaining all bird species composition and their groups in Mount Kenya forest reserve. However, the natural forest (undisturbed and disturbed) although showing some differences in overall species and forest dependency composition (i.e., because of more variability of species composition in undisturbed forest), they largely overlap in the NMDS illustrations (Figure 6.2). They also show low to moderate overall dissimilarities as shown by SIMPER

analysis (Tables 6.5-6.15), revealing that they have some similarities. The reason for the presence of greater overlap in species composition between the two forests is due to the mutual composition of more true forest birds (e.g., FS and FG (e.g., Bennun et al., 1996)) (see Figure 6.1b) or the presence of most insectivores and frugivores (see Figure 6.1c), that mainly depends on natural forests (e.g., Bennun et al., 1996; Şekercioḡlu et al., 2002 b; Mulwa et al., 2021). The components of ATHB restricted range species also include true forest birds that are characteristic to natural forest, majority of them being forest specialists of mainly insectivores and frugivores dietary guilds. The reason for presence of all these within disturbed and undisturbed forests is likely due to 'naturalness' shared between disturbed and undisturbed forests. This include similar structural attributes and resources that are generally beneficial to local birds and that serve to provide positive influence to birds. For example, (despite inherent differences in disturbance levels) the forest cover characteristics, foliage structural arrangement and complexity in both disturbed and undisturbed natural forests, are some of the beneficial forest structures for birds (e.g., Douglas et al., 2014). The resources can include food provision as host for preys, fruits and nectar compositions, and seeds and grains that serve to provide forage for different birds. High numbers of insectivores in natural forests (see Figure 6.1 c) could also be due to presence of high diversity and biomass of arthropods on which many insectivore birds depend on (e.g., Helden et al., 2012).

The existing difference between undisturbed and disturbed forests in terms of birds' species composition characteristics to each can be attributed to difference arising from local disturbance. It was shown in Chapter 4 that there are differences in habitat structural characteristics between undisturbed and disturbed forest, attributed to the local disturbance. This could probably have led to dissimilarity in resource availability for birds between the two forests affecting the species composition between the two forest types. For example, undisturbed forest is preferred by most forest specialist that are characteristic to it that belong to insectivores' and frugivores' dietary group than to disturbed forest. This reiterates the importance of undisturbed forest for forest specialist bird's survival in tropical montane forest, as well as how maintenance of these forests could maintain frugivore and insectivore bird species that can continue providing ecological services for the health of the forests. Although it was unexpected in this study

to find more frugivores characterising disturbed forest, other researchers have given possible explanation supporting this. For example, Gomes et al. (2008) say that presence of frugivore species in disturbed forest indicate the possibility that they can tolerate some form and level of disturbance in tropical forests.

The smaller number of forest specialist birds (and a smaller number of insectivores recorded) characteristics to disturbed forest relative to undisturbed forest, could be because of disturbance activities that removes preferred resources or conditions, that makes it unsuitable for forest specialists to occupy. Disturbed forest also is characterised by granivore species mostly belonging to FG category. This could be because of disturbance that encourages more canopy openings and growth of the grain plant materials such as grasses that may attract granivores. Comparable results were found in central Kenya highland agricultural landscapes (Ndang'ang'a et al., 2013), and in Cameroon (Tchoumbou et al., 2020).

The sharing of bird species composition between disturbed and eucalyptus plantation (that mostly belong to generalists, forest visitors and /or non-forest insectivores) means that the two forests share some habitat structural qualities. In Chapter 4, it was found that disturbed forest is characterised by high number of fallen trees, and high percentage of vegetation cover between 1 to 5 m above ground, making it have some structural similarities with eucalyptus plantation (i.e., of open canopies). These characteristics of disturbed forest might have therefore attracted similar species of birds to those in eucalyptus plantation such as insectivores mostly belonging to FG, FV, NF insectivores as shown in Table 6.12, and that were found to have some overlaps in NMDS illustration in Figure 6.3 c. Disturbance therefore contribute to simplification and opening of natural habitats, making it have similar characteristic qualities to exotic plantation and thus attracting similar bird species composition.

Nectarivores species (all forest generalists), were characteristic to both disturbed and eucalyptus plantations (Table 6.13). This is contrary to the expectation of this study. It was expected that natural forests (either undisturbed or disturbed) could have more composition of nectarivores than eucalyptus plantation owing to likely high variety of

flowering plants in natural forests (especially canopies) for foraging (e.g., Raman and Sukumar, 2002; Quesada et al., 2003). However, the nectarivores were found to characterise disturbed and eucalyptus plantation, likely owing to the presence of more flowering plants mostly from shrubby and herbaceous vegetations that characterised the two forest types. Within eucalyptus plantation, the shrubby and herbaceous vegetation were native, and the kind that associated with settlements and cultivated lands (personal observation). These vegetation grows and develop, encouraged by more light penetration in the eucalyptus plantation and disturbed forests (e.g., Feyera et al., 2002), and by developing flowers attracts nectarivores. These findings agree with what was found in Usambara mountains relating to the endemic Double-collared Sunbird (*Cinnyris usambaricus)* whose diet was found to mainly consist of non-native plants found mainly in areas with limited crown cover (e.g., Lueder, 2018). Similarly, Calvina-Cancela (2013) also found that birds that use flowers as an important food resource are mainly associated with shrubs at young age and with canopies at older ages in Eucalyptus plantation. In tropical montane environment of Ethiopia, nectarivores composition were found to be more in agricultural lands than forests (Gove et al., 2013).

The species composition in the eucalyptus plantation is different from those in undisturbed and disturbed forests. This is shown through the highest overall dissimilarities (of > 70 % in Tables 6.5 to 6.15) between any pair of natural forests (undisturbed/disturbed) and eucalyptus plantations and further illustrated by NMDS ordinations, showing almost separating or completely separating eucalyptus plantations from the other forest types (Figures 6.2-6.4). The structural characteristics of eucalyptus plantation that include its characteristics open canopy cover, tallest trees, and greatest tree to tree distances (dispersed) could be responsible for providing more resources and conditions attractive to FV, NF, granivore, nectarivores, and omnivore forest generalists, ATHB forest generalists, and ATHB insectivore species characteristics to eucalyptus plantation. Open canopies could be responsible for providing more opportunities for insectivore birds to forage and capture arthropods, and for grasses to develop in eucalyptus plantation floors that attracts more granivore bird species. Also, following disturbance and manipulation of plantation forests by humans, it might have led to presence of possible different variety of food sources that can be utilised by omnivores

and insectivores. For example, Ndang'ang'a et al. (2013) also found that omnivores' species richness was influenced positively by the increasing fallow and cultivation, conditions that may influence availability of variety of food sources suitable for omnivore bird species. On the other hand, the insectivores found here could be using different foraging strategies or utilising different food resources from the one found in natural forests, hence potentially found only in eucalyptus plantation. For non-true forest birds (especially FV and NF), generalists and associated dietary guilds, eucalyptus plantation forest seems to play an important role in maintaining the species composition of ATHB, especially those whose diet consists of insects. Non-forest birds and granivores were also found to characterise agricultural landscape in tropical montane forest of Ethiopia (e.g., Gove et al., 2013). The presence of these species in eucalyptus plantation in Mount Kenya, shows the potentials of exotic plantation to provide additional habitats for these birds, including the ATHB restricted species. Also, the presence of forest visitors and non-forest birds shows that eucalyptus plantation has a potential to provide a refuge and safe sites for these birds, especially when they are losing their suitable habitats elsewhere in settled, grasslands and farmlands. It is therefore evident that eucalyptus plantation forests in Mount Kenya are not biological 'deserts' (e.g., Lindenmayer et al., 2003) but can contribute as additional or complementary habitat, where particularly non-forest birds can thrive, or for other form of resources uses such as nesting, foraging or simply for cover. Matuoka et al., (2020) attribute the likely existence of few or no true forest birds in plantation forests to be because of simple structural characteristics that are associated with management practices. Douglas et al. (2014) also argued that exotic plantations exert little positive influence on forest birds, possibly because they offer poorer resources in terms of foraging, nesting, and shelter.

6.4.3 Conservation and management implications

The study has highlighted conservation values of each forest type and combined importance of all forest types as critical for overall conservation of birds' community composition found in Afrotropical montane areas. Other researchers have emphasised the importance of natural forests for conservation of local biodiversity (Brockerhoff et al., 2008; Brockerhoff et al., 2017), highlighting irreplaceability of it, particularly undisturbed

forest for specialised forest birds (Dayton, 1990; Gibson et al., 2011; Gilhen-Baker et al., 2022). This study also empahsise the role of combined natural forests, irrespective of local disturbance to be critical in conservation of forest birds in Mount Kenya, particularly for true forest birds that consist of FS and FG, including insectivores and frugivores dietary guilds.

Of importance is eucalyptus plantation emerging as the man-made forests contributing to the presence of some forest and non-forest birds of varied dietary guilds, and species restricted to Afrotropical highlands biomes. Past studies had reported mixed findings on ecological contribution of eucalyptus plantation in Afromontane forests (e.g., Bayle, 2019; Debie and Anteneh, 2022; Seifert et al., 2022), but this study is reporting it as new emerging habitat with unique habitat characteristics, that holds slightly unique bird species from those in natural forests, For example, it seems to have attracted FG and FV also found in natural forests, but have 'pulled' non-forest birds from non-forest habitats also. Therefore, although it is not known the usefulness of other surrounding land uses (i.e., farmlands, settled areas etc) relative to eucalyptus plantation in the current study, each forest type should be targeted by managers for conservation to ensure a complete protection of bird community found in Afromontane forests. For example, eucalyptus plantation has at least demonstrated its usefulness for FVs, NFs, insectivores, as well as granivores. It is likely that eucalyptus plantations offer resources for non-true forest birds, especially for insectivores (likely those gleaning for insects through or above the plantations) and granivores and therefore being part of ecosystem only offer its suitability just to that extent. Eucalyptus plantations, despite being an exotic tree plantation, also maintain species composition of ATHB species especially for generalists and forest visitors' insectivore group. Therefore, being established in the study area will only offer resources for generalists and forest visitors' insectivore and non-forest birds, in addition to other commercial economic benefits that eucalyptus is associated with.

To the forest managers and conservationists, based on the findings in this research, as a caution following the existence of eucalyptus plantation or severely disturbed forest types in the study areas, and especially in areas bordering farmlands, the likely increase in non-true forest birds (FV and NF) may prove risky. For example, the increase in the

occurrence of FV and NF and possibly more granivores, omnivores and insectivores' birds may saturate the composition within the area, with the presence of granivores and omnivores may increase possibility of destructive birds as pests in the nearby agricultural farms.

6.4.4 Conclusion and recommendation

Exotic plantations in lower montane part of Mount Kenya partly complement but do not replace natural forests in its bird composition. However, it has more species that belongs to FV and NF, insectivores and granivores than natural forests owing to having conditions reflective of both forest and non-forest ecosystems. Natural forests (undisturbed and disturbed) are particularly critical for more specialised birds such as forest specialists and Afrotropical highlands biome species, and those that are likely to play critical ecosystem services such as insectivores and frugivores. The combined forest types therefore bring together a community of birds present in Afromontane areas, with eucalyptus plantation attracting birds from non-forest habitats, probably by creating refuge and easing competition pressure of these birds in farmlands and human settlement sites.

Natural forests represented by undisturbed and disturbed forest types, differs in terms of habitat characteristics because of local anthropogenic disturbance (Chapter 4). However, they largely share some similarities in terms of bird species compositions that probably reflect the retained similarities in habitat characteristics that are left after local disturbance. Eucalyptus plantation on the other hand, contain composition of birds that also reflect its habitat characteristics relative to undisturbed and disturbed forest type. For example, it forms an important forest type for granivores, nectarivores and omnivores dietary groups of FGs, FVs and NFs attracted here probably by habitat characteristics identified in Chapter 4. Eucalyptus plantation, however, have little influence on the composition of true forest birds, particularly forest specialists than does natural forests (undisturbed and disturbed). Similar findings have been recorded by other researchers (e.g., Proença et al., 2010) that have also emphasized that forest species patterns may be affected by forest naturalness. Studies that have looked at the bird species composition in isolated eucalyptus plantation (i.e., that are removed from nearby native forests) have

found depauperate species in the plantations as compared to surrounding land uses (e.g., Phifer et al., 2017). But more research is recommended within the Afromontane forests on the status and species composition of forest and non-forest birds in eucalyptus plantations neighbouring natural and those far away from natural forests and if indeed the FV and NF birds found in eucalyptus plantation are the one typical of other non-forest habitats such as grasslands, farmlands or settled sites.

There were generally poor species richness and composition of nectarivores recorded in this study. However, there was unexpected higher composition in disturbed and eucalyptus plantation of nectarivores' FG than observed in undisturbed forests despite undisturbed forest potentially offering more flowering opportunities. Further studies of species composition and distribution of nectarivores is recommended in the Afromontane forest sites. This is to further understand their habitat requirements and particularly how it associates with natural forests and eucalyptus plantations, and how their distribution may affect their potential ecological functions such as pollination.

Natural forests (both disturbed and undisturbed) were particularly important for ATBH species compositions, especially for forest specialists and generalists (true forest birds) belonging to either frugivores or insectivores' dietary guilds. However, introduced eucalyptus plantation play a role also by having the species composition of ATHB forest generalists' and forest visitors' insectivore birds, making it a complementary habitat for these species. It is recommended that specific studies be done on insectivore bird composition in natural forest and eucalyptus plantation in Mount Kenya, to determine if this dietary group differs in their feeding strategies in different forest types.

Chapter 7- General discussion

7.1 Introduction

Tropical Afromontane forests continue to play a critical role in providing socio-economic, cultural, and recreational benefits to an increasing population of people, particularly those surrounding the forests (Mengist et al., 2022). At the same time Afrotropical forests are key areas to conserve and maintain other biodiversity dependent on forests, as well as for ecological functions such as climate regulation, flood and soil erosion control, watershed protection and as key sources of water. This research arose from the observation that Afromontane forests are currently managed using approaches such as participatory forests management (PFM) that allows local communities to access and use forests as part of their involvement in conservation. Among other things, PFM engages communities in the management of forests by sharing with them benefits accrued from forest resources (see Himberg et al., 2009; Mbuvi et al., 2009). This approach has been in use without much empirical justification on how it impacts biodiversity conservation in tropical forests. It is well known that human use of forests introduces disturbances that affect both vegetation habitat characteristics and other organisms dependent on forests (e.g., Smiet, 1992; Morris, 2010; Majumdar and Datta, 2015; Popradit et al., 2015; Alroy, 2017). But there is no research so far that has tried to assess how the use of forests and associated disturbances, have changed habitat characteristics, and how the disturbed forest area differs from the undisturbed forest (i.e., as two different forest types). In addition, exotic plantations are widespread across the Afromontane landscape, particularly used to rehabilitate degraded sites and to increase forest cover in formerly non-forested parts of the montane region. Most of the exotic plantations are established close to the existing natural forests as part of the landscape matrix. However, there is no study that has tried to compare inherent characteristics of the exotic plantations to that of the natural forests, yet exotic plantations potentially introduce characteristics that are unique, such as new vegetational structures in the ecosystem.

The ecological role of disturbance is well known in tropical forests (Attiwill, 1994; Denslow, 1995; Alroy, 2017; Gray et al., 2018; Burton et al., 2020), but specific effects

of local disturbance on organisms mediated by habitat structural change in Afromontane forests needs further assessment. Ecological and conservation values of exotic plantations are still contested in Afromontane areas and further information on this is needed. Furthermore, there is a need to determine the contribution of different forest types (undisturbed, disturbed and exotic plantation) on bird diversity and community composition in Afromontane forest areas.

This thesis aimed at determining the habitat characteristics variation within and across undisturbed and disturbed forest types (resulting from anthropogenic forest disturbance), and in exotic eucalyptus plantation, and bird species richness and abundance and community composition in these three forest types in Mount Kenya's forests. Using bird and habitat structural characteristics data collected over a year in a total of 190 point counts distributed in the three forest types, across three study sites, in eastern, southeastern, and southern Mount Kenyan forest, the resultant findings provide important new information on habitat characteristics, bird diversity and community composition of the forest types. This chapter synthesises these research findings highlighting the influence of local anthropogenic disturbance on forests and conservation values of each forest type. This chapter also discusses conservation and management implications of the findings and highlights opportunities for future research.

7.2 Synthesis of key results

In the Afrotropical region, studies have looked at the contributions of PFM to local communities (e.g., Schreckenberg and Luttrell, 2009). Most have also compared how PFM have improved forest conservation relative to areas without PFM (e.g., Blomley et al., 2008; Matiku et al., 2012; Ameha et al., 2016; Tadesse et al., 2016; Kairu et al., 2021). However, to the best of the current understanding, none have compared how PFM changes forest habitat structural characteristics in areas where humans access and harvest forest resources and in areas where they do not (i.e., disturbed, and undisturbed forest sites). This is in addition to the role of illegal anthropogenic activities that disturb forests and change structural characteristics that have not been extensively studied in Afrotropical region (but see Borghesio, 2008; Mammides et al., 2015), unlike in other tropical regions

(e.g., Thapa and Chapman, 2010; Morris, 2010; Naveen et al., 2021). Although it is known that exotic plantations and natural forests are inherently different based on their growth characteristics and vegetation structures (Brockerhoff et al., 2008; Capossoli et al., 2009; Nichol and Abbas, 2021), vegetation characteristics that are similar or dissimilar across these forest types is unknown. The characteristics within the exotic plantation that are not similar to those found in natural forests, are potentially new to the landscape where the plantation is part of the matrix, and its likely role in conservation of native species is unknown.

This research has demonstrated that the local anthropogenic use of Afromontane forests has some negative impacts on natural forests' vegetation structural characteristics by reducing complexity while exotic plantation introduces new habitat features. However, local disturbance in natural forest enhances bird species richness, abundance and changes community composition of some birds and negatively affect others within disturbed forest. Specifically, the findings in Chapter 4 shows that all three forest types vary significantly from each other, with variation significantly different in 18 out of the 23 (78 %) of the habitat characteristics measured across the forest types. Whilst this difference is not unique across the forest types and it is expected given the nature of the forests (e.g., Hitimana et al., 2004; Borah and Garkoti, 2011; Alroy, 2017; Teucher et al., 2020), the fact that the forest types arise following local scale forest disturbances on natural forests and introduced exotic plantation as a new forest type, flags up the existence of forest types based on easily overlooked circumstances such as local disturbance occurrences, that is enhanced through forest management supported policies such as PFM. The forest types thus arise and exist as entities whose classification as undisturbed, disturbed and plantations becomes concepts suitable to be subjected to scientific investigation on their valid usage and application in the study area.

Within each forest type across sites, undisturbed forest had 18 out of 23 habitat characteristics with significant variation within it, most of these based on vegetation structures (Table 4.3). In addition, among the significantly varied habitat characteristics within undisturbed forest, nine (9) of them namely: tree DBH, basal area, % vegetation cover at 5-8 m, % vegetation cover at greater than 8 m, % canopy cover, litter depth and

number of snags all had highest means in undisturbed forest type than any other forest. Based on percentage indicator values (IndVal), the same habitat features (tree DBH, basal area, % vegetation cover at 5-8 m, % vegetation cover at greater than 8 m, % canopy cover, litter depth and number of snags) characterised undisturbed forest (Table 4.7). These shows that undisturbed forest is structurally varied in terms of vegetation characteristics which makes the site more structurally complex. This complexity is reduced in a disturbed site where only 10 out of 23 habitat characteristics differed significantly within it (Table 4.5), and at the same time only 4 habitat characteristics (% vegetation cover at 3-5 m, foliage height diversity (FHD), number of saplings, and number of fallen trees) had highest means measurements than other forest type (Table 4.2). Percentage vegetation cover at 3-5 m, foliage height diversity (FHD), number of saplings, and number of fallen trees was also identified to have significantly high indicator values for disturbed forest type (Table 4.7). These findings suggests that undisturbed forest is structurally complex (having many significantly different structural parts/characteristics) while disturbed forest is structurally less complex (have few significantly different characteristics), but diverse based on foliage arrangements. According to these findings, the level of local forest disturbance i.e., through PFM arrangement or illegally reduces forest complexity in disturbed forest type in Afromontane forests while it has increased forests foliage structural diversity in disturbed forest (Chapter 4). Studies have identified undisturbed forest as complex (e.g., Bawa and Seidler, 1998; Van Gemerden et al., 2003; Hitimana et al., 2004) and have also blamed disturbance on reduction on this complexity (Morris, 2010; Alroy, 2017; Jara-Guerrero et al., 2021). Since the time of MacArthur and MacArthur (1961), foliage diversity has been noted for increasing species diversity, and it seems to be the case even in the recent studies in tropical forests (Jayson and Mathew, 2003) including this study in Afromontane forest.

Only 6 out of 23 habitat characteristics within eucalyptus plantation varied significantly, leaving most characteristics to signify a homogeneity of vegetation structures within the plantation. This is expected for the plantation given the planting, growth characteristics and silvicultural practices in the plantations. Unique habitat characteristics to eucalyptus plantation are tallest trees, dispersed trees, high percentage of vegetation cover close to forest floor (i.e., at 0-1 m) and number of cut trees and human trails (Table 4.7). Some of

these characteristics are particularly new features in Mount Kenya montane forest (and probably elsewhere in the Afromontane where eucalyptus plantations are established). These potentially create new habitat opportunities for other organisms or complement the natural forests (undisturbed and disturbed) in supporting native species, i.e., as does emerging forests in urban areas (Kowarik et al., 2019). As one of the first studies to compare habitat characteristics across existing forest types including exotic plantation in a landscape like in this study, the findings in Chapter 4 provides novel illustration/snapshot of how habitat structural characteristics are arranged within the landscape and fills an important knowledge gap in Afromontane forest ecosystem. The arrangement of these habitat characteristics, together with how it supports the local taxa, can help in decision making by managers and conservationists on how to enhance local biodiversity or how to address the associated challenges affecting the local biodiversity in the study area. It also provides important information to forest managers and conservationists on existence of variety of habitat structural characteristics in existing and emerging forest types, and possibilities of designing ways to enhance or reduce these varieties to increase opportunities to support other local forest dependent taxa. The subsequent chapters establish how forest types and habitat characteristics within it drives bird species richness and abundance and community compositions.

Chapter 5 of this thesis investigated the conservation values of forest types observed in Chapter 4, and how specific habitat characteristics influenced bird species richness and abundances. Birds are used to assess conservation values of forest types because they are sensitive to changes within their habitats (Sodhi et al. 2005; Kumar and Shahabuddin, 2006; Yap et al., 2007; Wu et al., 2011) and can respond faster to disturbance (Ramírez-Soto et al., 2018) than other taxa. For the management purposes and conservation decision making at the study area and other related areas, the findings in Chapter 5 revealed that each forest type plays a significant role for species richness and/or abundance for bird groups considered except for forest generalists' (FG's) and nectarivores' species richness, and insectivores' species richness and abundance. This reveals that the level of local anthropogenic disturbance in Afromontane forest of Mount Kenya, apart from causing changes in habitat characteristics seen in Chapter 4, also indirectly causes noticeable change on diversity of other forest dependent organisms (e.g.,

birds richness and abundance in this case) that are dependent on the studied habitat characteristics. The management should therefore notice that the level of disturbance subjected to disturbed forest type by the local communities, either illegally or legally has potentially increased foliage-based vegetational characteristic diversity, that has likely significantly increased conditions and opportunities for more bird species richness and abundance, and for those species restricted to Afrotropical montane forests. Although this was unexpected given that most studies have always reported less species richness and abundance following disturbance in tropical forests (e.g., Waltert et al., 2005; Munyekenye et al., 2008; Alroy, 2017; Mulwa et al., 2021), Asefa et al. (2017) had reported an increase in species richness and abundance in disturbed unprotected forest in Ethiopian highlands. This means the current level of local community disturbance in natural forest in Afromontane, i.e., through selective logging and harvesting of forest products, that are mostly encouraged and controlled by participatory forest management (PFM) approach, is probably not high enough to degrade forest and cause reduction in species richness/abundance of birds. Instead, the management of forest in the study area might need to understand that the existing local disturbance might be creating opportunities for more widespread species (e.g., forest generalists) or those attracted by disturbed conditions (forest visitors and/or non-forest birds) to occupy the site, raising both overall species richness and abundance. The recorded high foliage height diversity in the disturbed forest type (Chapter 4) might be because of this local community disturbance, and these may not reflect forest complexity (with high-quality) forest type as undisturbed forest do but may only reflect the occurrence of additional niches (heterogenous structural foliage arrangement) that accommodate more species or abundance. Therefore, according to this study, the forest management and conservationists will thus need to know that a high species richness or abundance of birds in the study area, is not necessarily a suitable indicator of forest quality/complex forest in the current study. To determine the quality of forest type in the study area may need determination of specific specialised species that are characteristics to a given forest type.

It was established in this study that undisturbed forest in Afromontane forests is important for forest specialists (FS), species richness of frugivores and of Afrotropical highlands biome restricted range species (ATHB), Kenya mountains endemic bird (KMEB) and one

threatened, *Cinnyricinclus femorali*) (Vulnerable). For example, habitat characteristics related to undisturbed forest type e.g., canopy cover, % vegetation cover at > 8 m, number of snags, altitude and distance from forest edge also significantly and positively predicted these species groups. Based on these characteristics, for management and conservation purposes, this research suggest that undisturbed forest is irreplaceable by any other existing forest type in Afromontane forest. This is because the species recorded in and predicted by undisturbed forest related characteristics are more specialised forest birds that require intact undisturbed forest (Bennun et al., 1996; Newbold et al., 2013; Deikumah et al., 2014; Menezes et al., 2016; Mulwa et al., 2021) and it is not surprising to have more species richness and/or abundance in undisturbed forest type. Asefa et al. (2017) also found high species richness and abundance of FS in protected forest in Ethiopian Afromontane forest. This study found insectivores' richness and abundance to be not significantly different across forest types, but it was variously predicted by different habitat characteristics. For instance, insectivores' species richness was positively predicted by % canopy cover, and altitude associated with undisturbed forest, yet also positively predicted by % vegetation cover at 0-1 m associated with eucalyptus plantation. From this, the thesis highlights the importance of finer details of species such as sub-groupings. In other words, insectivore birds are found in all forest types, but they have been considered in a very broad group within which are different species with different feeding strategies (i.e., arboreal, understory, aerial, and foliage gleaners). Grouping them together masks those specific species associated with a specific forest type.

Eucalyptus plantation recorded the second highest overall bird species richness (after disturbed forest type) (Table 5.4) which is attributed to mostly forest visitor birds (FV), non-forest birds (NF) and granivores that had the highest species richness and abundance in it than other forest types. The unique habitat characteristics to eucalyptus plantation (tallest trees, dense vegetation cover at 0-1 m, and dispersed trees) positively predicted species richness of FG and insectivores, and species richness and abundance of FV, NF and granivores. However, some FS (8 species) was surprisingly recorded in eucalyptus plantation (see Figure 5.5 for an example). This was unexpected as some studies have reported poor conservation values of eucalyptus plantations than natural forests (Goded et al., 2019; Lemessa et al., 2022) while others have reported mixed reactions for

conservation contribution of exotic plantations (e.g., Barlow et al., 2007 (b); Norton, 1998; Brockerhoff et al., 2008; Farwig, et al., 2008). From these findings, it is thus evident that, apart from FGs, eucalyptus plantation attract species that do not normally occur in forests (i.e., FVs and NFs), therefore eucalyptus could probably be acting as an important refuge site for grasslands, agricultural and other non-forests sites bird species, attracting them closer to forest ecosystems. However, eucalyptus plantation may also act as temporary refuge sites for specialised birds like FS, using it as just temporary habitat or refugia. It is probably not very useful to these specialised forest birds since none of the characteristics of eucalyptus plantation positively predicted these species. For management and conservation purposes, the findings in this study thus puts eucalyptus plantation in Afrotropical montane forest as serving the following: (i) creating important refuge sites for non-forest birds owing to creation of conditions and possibly resources found in non-forest sites (ii) offering similar/comparable or additional suitability for forest generalists and some insectivores relative to other forest types (iii) can offer temporary refuge to specialised birds such as FS albeit in lower diversity compared to natural forests (iv) are suitable sites for granivore species richness and abundance.

No study has directly compared bird species composition across and between undisturbed, disturbed and eucalyptus plantation forest types in Afromontane region. After assessment of the bird diversity and how habitat characteristics predict species richness and abundance in Chapter 5, Chapter 6 examined community composition across forest types. The occurrence of most of the shared species' composition, and ANOSIM's significant differences across all forest types revealed in this chapter, affirms that all forest types existing in Afromontane forest of Mount Kenya, irrespective of local disturbance and exotic nature of eucalyptus plantation, together plays an important role in bird conservation. However, the role played by eucalyptus plantation can only serve to benefit mostly non-forest birds as it likely attracts these from the surrounding non-forest habitat matrix. The most benefiting bird species group from the presence of these range of forest types in Afromontane are forest generalist and insectivores. This could be the case given that generalists' birds can use different habitat types while insectivores could be utilising their range of different foraging strategies (e.g., aerial gleaners, foliage gleaners, bark foragers (e.g., Castaño-Villa et al., 2019)) and feeding stratum offered by different forest

types and therefore these species occur in all forest types. The Chapter provides further evidence that the presence of additional forest types (i.e., disturbed and eucalyptus plantation) within the landscape provides additional opportunities to local bird populations, especially FV and NF birds. Chapter 6 has also demonstrated that the existence of exotic eucalyptus plantation provides potentially useful additional sites for Afrotropical highlands biome restricted-range species, particularly non-forest dependent ones, removing possible assumption that restricted-range birds can only be found in native ecosystems or habitats where they are naturally suited or have evolved in.

Based on exclusive species found within each forest types, as well as demonstrated by NMDS illustration and SIMPER analysis, Chapter 6 serves to emphasise the conservation value of natural forests in Afromontane forests for forest dependent birds (FS and FG), that consisted mostly of insectivores and frugivores. That is, as shown in Figures 6.2-6.4, and Tables 6.5-6.15, there are generally more NMDS overlaps and low overall SIMPER dissimilarities between undisturbed and disturbed natural forests for all bird groupings (Tables 6.5-6.15). This agrees with findings of other researchers (e.g., Dayton, 1990; Gibson et al., 2011; Gilhen-Baker et al., 2022), that the natural forests, especially the undisturbed forest is irreplaceable. From management and conservation point of view therefore, this fact presents the greatest challenge currently as most undisturbed forest are always under threat of disturbance following the growing human population needs in Afromontane areas (Teucher et al., 2020; Razgour et al., 2021). In addition, these findings shows that the species that are in greatest risk of being locally lost in Afromontane forests following local disturbance are forest specialists, consisting mainly of insectivores and frugivores. This is because the resultant disturbed forest arising from PFM activities and the presence of eucalyptus plantation in Afromontane forests benefits mainly widespread species like FG, and FV, NF and granivores with some insectivores, nectarivores, omnivores, carnivores and ATHB restricted range species. It is also evident from this study that the presence of disturbed and eucalyptus plantation pulls the species not normally found in forests to these forests, probably acting to reduce competitive pressure on surrounding farms and settled areas, and reduction of known bird pests to crops such as the speckled mousebird (*Colius striatus*), Brimstone canary (*Serinus sulphurata*), Streaky seedeater (*Serinus striolatus*) or weavers observed in these forest types.

Despite a high dissimilarity (of > 70 % in Tables 6.5 to 6.15) in bird species composition between eucalyptus and disturbed forest types (also illustrated by almost separating/completely separating NMDS ordinations in Figures 6.2-6.4), this study revealed a possibility of local anthropogenic disturbance to contribute to certain similarities between disturbed and eucalyptus plantation forest types. For instance, disturbed and eucalyptus plantation shared some similarities in habitat characteristics related to disturbance, particularly open canopy cover. This could have contributed to the two forest types sharing bird species belonging to FG, FV and NF, composed of mainly insectivores, but the species that were characteristic to both are nectarivore bird species which were all forest generalists (Table 6.13). Based on these findings therefore, forest managers should know that local disturbance could be opening canopies for light penetration that encourages flowering shrubs and herbaceous vegetation that characterised the two forest types and thus attract nectarivore species.

Chapter 4 demonstrated eucalyptus plantation as a simple forest with higher homogeneity of habitat characteristics. Chapter 6 on the other hand has highlighted a highly dissimilar species composition of eucalyptus plantation with those of other forest types. Eucalyptus plantation is significantly characterised by bird abundance belonging to overall bird species, species of ATHB groups of either FG, FV, NF, and being mostly insectivores, granivore, nectarivores, and omnivore. Therefore, eucalyptus plantation plays a role in attracting widespread FG, FV, and NF bird species, that can also potentially act as important pollinators, crops pests, or they can also reduce insect pests within the ecosystem they occur. Eucalyptus plantation therefore do not assist natural forest types in bird species composition, and specifically undisturbed forest, but it just occurs only as an additional forest type that only attract mainly non-forest bird species from the non-forest surroundings within a landscape where it is also currently a part of its matrix.

The findings of Chapter 6 of this research can be concluded with the following:

 (i) Forest types in Afromontane area play a complementary role for shared bird species

For management purposes, despite the role played by each forest type in species richness and abundance of forest dependency, dietary, and Afrotropical highlands biome restricted

species, all forest types complement each other in terms of shared bird species composition. That is, they share more species richness of overall birds than any individual forest type or those shared between any paired forest types. Together, these forests contribute positively to overall bird composition in Afromontane region by probably providing opportunities in a complementary manner required by majority of birds. By doing these, forest types may spread the birds probably reducing likely competition between them (e.g., Jankowski et al., 2012). More importantly, there is complementarity of species composition within natural forest types that is of interest. For example, the higher number of forest bird's species composition shared between the paired natural forest types reveals combined importance in species composition in Mount Kenya montane forest. For instance, undisturbed and disturbed forest both shared 33 species of mostly forest specialists and generalists, yet each one uniquely holds less species, 4 and 16 species respectively (Figure 6.1 a). In addition, both undisturbed and disturbed forest types shared most of the species' composition of highlands biome restricted species, ATHB, which are more than shared between either of undisturbed/disturbed and eucalyptus plantation. This makes eucalyptus plantation to be less of conservation importance relative to the three forest types in Afromontane forests.

(ii) Irreplaceability of natural forests in Afromontane areas

The existence of most shared species of FS, FG, insectivores, frugivores and species of Afrotropical highlands biome between undisturbed and disturbed forest types (both natural), could be in response to some similarities in habitat qualities and characteristics, or shared 'naturalness' in both natural forest types. But the reduction of FS, FG, insectivores, frugivores and ATHB between pairs of disturbed/undisturbed and eucalyptus plantation, shows a likely reduction in habitat qualities in eucalyptus plantation, and that dictates the dissimilarities of the paired forest types. This is also in terms of habitat structural characteristics differences and disappearance of 'naturalness' in eucalyptus plantation, that could otherwise serve to attract true forest birds (FS and FG). This is thus reflected in the number of shared bird species composition between disturbed/undisturbed forests (natural) and eucalyptus plantations (exotic). The absence of natural forest qualities in eucalyptus plantation removes the suitability for it to be occupied by the birds

that requires high quality habitats, leaving only undisturbed and disturbed natural forests to be occupied by those birds which are sensitive to forest quality degradation such as forest specialists, insectivores and frugivores (e.g., Bennun et al., 1996; Gomes et al., 2008; Gove et al., 2013). Thus, for conservation and management purposes, these findings emphasises the irreplaceability of natural forests, particularly undisturbed forest by any exotic plantations (e.g., Gibson et al., 2011).

(iii) Eucalyptus plantations are suitable only for widespread and non-true forest bird species composition in Afromontane tropical areas

For the sake of management and conservation of the studied forest and other related areas in Afromontane region, although the eucalyptus plantation studied were near natural forests and share some attributes and species composition with disturbed forests, the higher-than-expected overall species richness and abundance per point count in eucalyptus plantations is attributed to higher number of FV and NF (species that are not normally dependent on forests e.g., Bennun et al., 1996). In addition, overall dissimilarity percentage between any pair of natural forests (undisturbed/disturbed) and eucalyptus plantations in terms of species composition is much higher (> 70 %) and NMDS ordination shows generally separating eucalyptus plantation from other forest types. Most FV and NF recorded in eucalyptus plantation belonged to insectivore, granivore, and ATHB restricted range bird species. The same group formed most bird species that were unique to eucalyptus plantation forest type. However, the widespread species of FG recorded (also shared with disturbed forest type) were mostly nectarivores, whose abundance per point count in eucalyptus plantation were higher than any other forest type, probably attracted here by the observed native herbaceous and shrubby vegetation that grow extensively understory of eucalyptus plantations.

7.3 Pertinent conservation and management implications

After the implementation of participatory forest management approach (PFM) in Afromontane region, apart from how it benefits local communities and how it helps to curb forest loss (sustainable use) (Matiku et al., 2012; Okumu and Muchapondwa, 2020), the question had remained how the local use of the forest impacts on biological

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conservation of the forests. This has been one of the main management constraints highlighted in section 4.3 (a) (i.e., lack of updated information) in the Mount Kenya Forest Reserve Management Plan (MKFRMP) (KFS, 2010). This also constituted a priority research area identified by Mount Kenya Ecosystem Management Plan (MKEMP) 2010-2020 (KWS, 2010). For example, under its Objective 4, action line number 4.8, on priority management-oriented research, MKEMP had identified ecological studies such as species-habitat interactions, community-forest interactions, and issues on reforestation as among 13 priority research areas required for the Mount Kenya ecosystem.

It has thus been established by this study that the local anthropogenic use of forests reduces forest complexity (in disturbed forest type) that negatively affects bird forest specialists, just as other researchers have established (e.g., Khanaposhtani et al., 2012; Leaver et al., 2019; Gumede et al., 2022). However, it can also enhance local diversity of foliage structural arrangements (Chapter 4) and serve to benefits some birds e.g., increasing species richness (see also Asefa et al., 2017; Leaver et al., 2019). The presence of Afrotropical highlands biome restricted species and increase in frugivores reiterate the importance of disturbed forest type even for species of conservation concern, and potential seed-dispersers within the forest respectively. Although, the general implication for this study is that it is beneficial to maintain the current level of local use of disturbed forest, and with proper management of eucalyptus plantation (i.e., ensuring that it has native vegetation growing within it, not harvesting early, reduced human traffics within it) could be more beneficial to wider local avifauna in Afrotropical montane forests. In addition, the management and conservationists would be interested to address forest simplification by controlling intensive resource harvesting in forest and any other on detrimental effects of local use that affect forest specialist species. They will also be encouraged to maintain the current forest use levels, either by setting up ways of reducing more destructive activities or maintaining the zonation schemes that control user intensities of different forest areas depending on conservation needs. In other words, stakeholders in forest management and conservation may enter into mutually enforceable agreements that define the sustainable use of defined forest resources, responsibilities, and authority in the management of the Afromontane forests.

The study established that forest specialists (FS) are positively predicted only by characteristics associate with undisturbed forest and negatively by disturbance related characteristics. In addition, FS and Afrotropical highlands biome restricted species (ATHB) are mostly shared within natural forests (undisturbed and disturbed). These are species that are in greatest risk of being locally lost in Afromontane following local disturbance. The management and conservationists should therefore be aware of the greatest importance of characteristics that signify the quality of undisturbed forest for forest specialists and natural forests in general for ATHB species. This is because these groups consist of more specialised species that cannot be conserved in eucalyptus plantation. Appropriate management decisions and planning for the conservation of natural forest is emphasized for the sake of the specialised bird, for example through enhancing protection of undisturbed forest structural qualities such as canopy covers.

Literature in forest ecology (see Chapter 2 section 2.5, sub-section 2.5.1) have highlighted in detail the importance of forest structure particularly the snags (dead standing trees) for other forest taxa (e.g., Cockle et al., 2011; Burgar et al., 2015; Seibold et al., 2015). In this study, although undisturbed forest was significantly characterised by highest number of snags (Chapter 4, Table 4.7), number of snags did not positively predict any bird species group except omnivores' species richness which was negatively predicted by number of snags. Disturbed forest had lower number of snags, attributed to observed felling of dead dry standing trees for fuelwood in the study area (See Figure 7.1). The illegally felled dead and drying tree in Figure 7.1 (a) has been cut into small logs in preparation for cutting into fuelwood as observed in disturbed forest type in the study area during this research. Following this, it prevents it from natural processes of rotting (Figure 7.1 (b) that turns it to more suitable habitats for other organisms hence denies suitable foraging sites for species that depends on arthropods as shown in Figure 7.1 (c) (see also Chapter 2, Figure 2.2). Although it had no significant positive effects on bird species richness or abundance in this study, for the sake of other organisms as was observed in Figure 2.2 and Figure 7.1, forest managers and conservationists should consider controls on the type of fuelwood allowed to be collected under PFM arrangement and that ensures protection of all snags in the forest types.
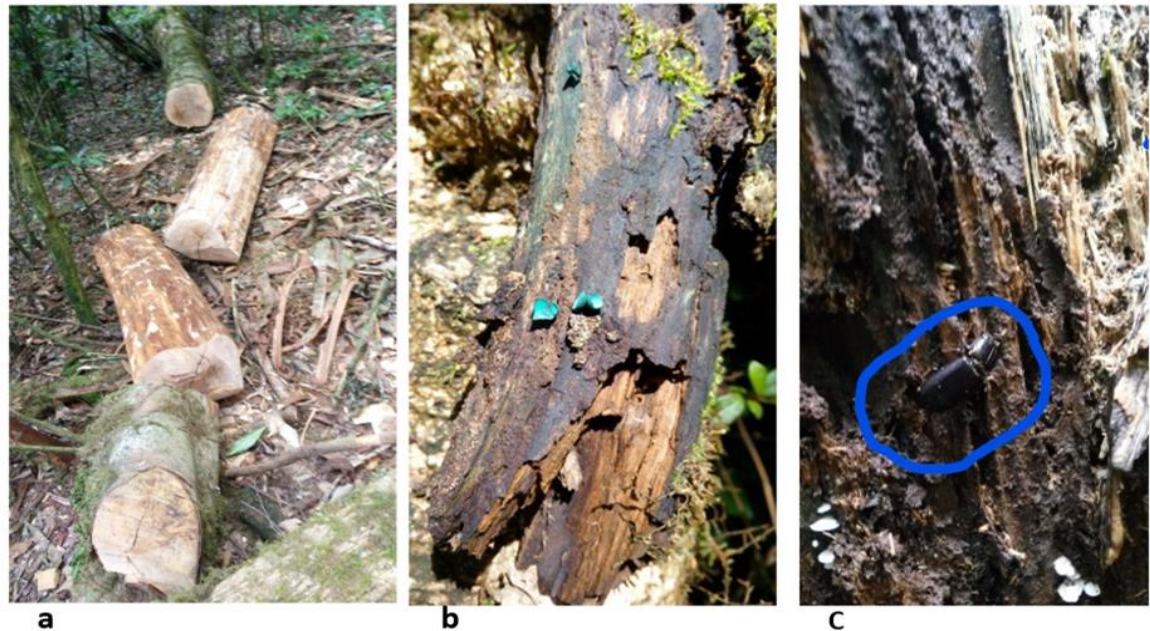
Figure 7.1: (a) Felled dry tree cut into logs for fuelwood in disturbed forest type in Mount Kenya forest, that removes it from rotting (b) and thus deny other organisms (e.g., circled in blue in (c)) suitable foraging habitats for species such as birds feeding on arthropods (insectivores). Photos taken on (a) 23/02/2019, (b) 22/04/2019 and (c) 08/05/2019 by author.

The established exotic plantation by Kenya Forest Service (KFS) management to rehabilitate degraded forest sites and to increase forest cover has also been proved through this study to be not that ecologically useful locally in Mount Kenya, and perhaps in other areas of Afromontane region. For example, eucalyptus plantation forests just attract only forest generalists (FGs) and forest visitors (FVs), and otherwise non-forest (NF) birds belonging to insectivores, granivores, nectarivores and omnivores probably from other non-forest ecosystems such as grasslands. In addition, they have been shown to host non-forest birds of Afrotropical highlands biome restricted bird species. By so doing, eucalyptus plantation only supports these birds, providing probably new habitat niches, easing birds' dependence on other non-forest habitats such as farmlands, grasslands, and reducing incidences and pressure of birds being crop pests in farms. However, eucalyptus plantation has no usefulness or proven contribution for true forest bird's species or those species that are ecologically useful to the environment. But given that the surrounding areas of Afromontane forests such as Mount Kenya are occupied by highly agricultural

communities, crop farms grown next to eucalyptus plantations at the edges of forests may suffer from granivore bird species crop pests, although they can also benefit from pollinators and those insectivores that feeds on crops pests as demonstrated by Milligan et al. (2016) and Tela et al. (2021). As observed, exotic plantation may serve to partly provide refuge to Afromontane biodiversity like FV and NF from destruction of their natural habitats such as grasslands (e.g., Barlow et al., 2007 (b)), therefore saving species that would have otherwise been lost due to lack of suitable habitats. Although the above is true, the findings shows that eucalyptus plantation cannot replace or be in place for natural forests, as species and their groups found in eucalyptus plantation can also be found in natural forests depending on their disturbance conditions, but not vice versa for specialised bird species. It was further observed that all eucalyptus plantation studied were characterised by native vegetation undergrowth. It is not known however, the conservation role of eucalyptus plantation without native vegetation undergrowth but it is recommended here that management be considered on eucalyptus plantation having the native undergrowth to provide suitable habitats for birds. Similar recommendation was also given by John and Kabigumila (2011) in East Usambara Afromontane hotspot. In summary therefore, it is the recommendation of this study that Afromontane forest management should first consider other native tree species (preferably the first growing species) for rehabilitation purposes, instead of exotic trees like eucalyptus plantations, so that the contribution of native forests is enhanced. The use of exotic plantation like eucalyptus does not add any new value, more than the one provided by any other non-forest ecosystem and therefore should not be used for rehabilitation in degraded forest sites within Afrotropical montane forests.

7.4 Opportunities for future research

This study demonstrated the change in habitat characteristics because of both illegal and legal use of forests by local communities living adjacent to the forest. The legal aspect comes because of local communities having been allowed to form group membership of community forest associations under the umbrella of participatory forest management. As members, under the negotiated agreement on how to use and conserve forests are allowed to benefit from forest resources such as beekeeping by placing their hives in forests,

livestock feed collection, fuelwood collection among other benefits. However, it was observed that where there was placement of beehives, removal of fuelwood and livestock feeds among other harvesting practices, are accompanied by physical changes in the structure of tree or vegetation involved, following the removal (e.g., Figure 7.2).
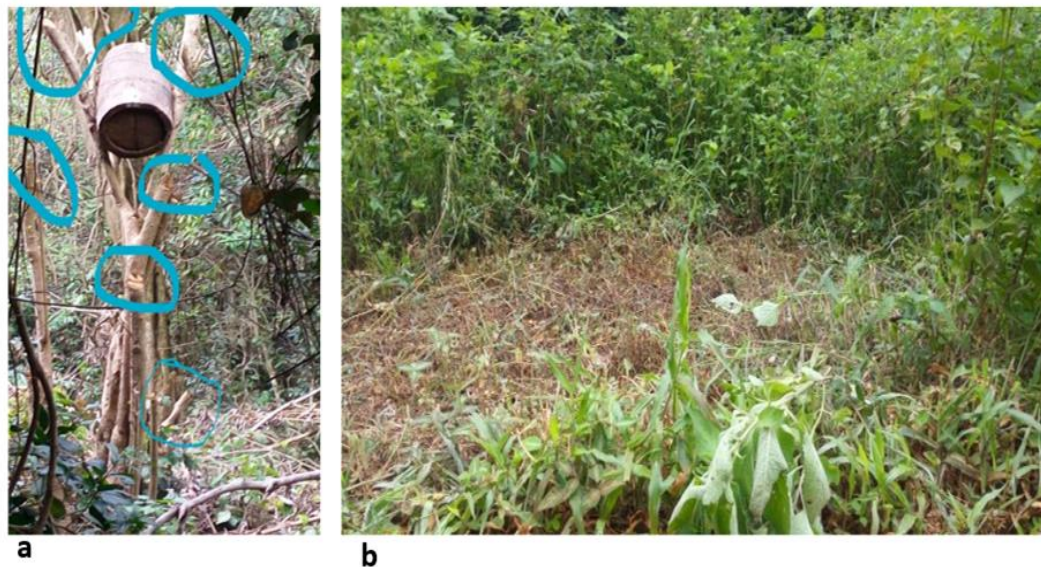
Figure 7.2: (a) Sites for beehive placement characterised by extensive vegetation cutting (circled parts with blue colour) and (b) sites for livestock feed harvesting cleared of vegetation in Mount Kenya. Photos taken on (a) 03/04/2019 and (b) 24/6/2019 by the author.

These practices potentially lead to substantial loss of vegetation (and vegetation cover) that would have otherwise been beneficial to provide forage, cover, nesting sites and habitats for other organisms. It would be informative to further assess how much of these vegetation covers (and biomass) that are likely to be lost through such removal practices, and to document possible amount of natural vegetation lost that would have been useful to other organism in the forest. This would inform managers and conservationists on strategies to adopt to address likely unsustainable use of forest under PFM and likely impacts on other organisms.

Natural forest was found to be particularly important for forest specialists. But unexpectedly, some forest specialists though negligible number was recorded in eucalyptus plantation (e.g., Figure 5.5). They might have been recorded by chance, or use eucalyptus plantation for temporary refuge, or they are attracted by native vegetation

growing underneath eucalyptus plantation trees and that raises eucalyptus conservation values, or combination of these. Therefore, in-depth, and focused research is required to reveal how and why these birds associate with exotic eucalyptus plantation.

Insectivore birds did not significantly differ across forest type in Mount Kenya forest reserve, and they were positively predicted by habitat characteristics associated with both undisturbed, disturbed and eucalyptus plantation in this study. This is attributed in this study to the lack of grouping insectivores into sub-groups associated with foraging strategies. It is therefore not clear how insectivores associate with forest types and habitat characteristics, and it is recommended that for future research on insectivores in Mount Kenya forest reserve, sub-groups such as understory insectivores, aerial/canopy, and foliage gleaners among other groups should be considered. Similarly, frugivores did associate with habitat characteristics for both undisturbed and disturbed forest types although there were significantly different across forest types. Future research is also recommended on frugivores that feed on large fruits and small fruits separately, as combining them together may mask how they associate with habitat characteristics related to a given forest type.

This study highlighted the importance of eucalyptus plantation for granivore species. Eucalyptus plantations play a host to granivores than any other forest type in this study. But granivores as highlighted earlier, may potentially play a destructive role in tropical Afromontane forests by removing or consuming any regenerating native forest seedlings, shoots, or seeds (e.g., Wright, 2003; Terborgh et al., 2018), or likely be destructive to nearby crop farms. This needs to be further investigated and determined especially on how likely the granivores can slow down or lead to poor or no forest re-establishment, and reduction on crop production in nearby adjacent farms in Afromontane forests.

The eucalyptus plantation studied comprised pockets of plantations next to the natural forests within the protected areas. This could have facilitated easy reach of both birds and dispersal of seeds from natural forest to eucalyptus, thus encouraging growing native vegetation underneath the plantations. Although, all efforts were made during the study as detailed in Chapter 3 to ensure that bird species recorded within each forest type were

independent from each other, it is possible the proximity of eucalyptus plantation to the natural forest have influenced easy movement of natural forest birds into the plantation and vice versa. This is because birds are highly mobile. A similar study is recommended where possible that ensures complete independence of forest types through considerable distances apart. It will also be informative to consider eucalyptus plantation with no native vegetation growing underneath to disentangle the effect of native vegetation on conservation values of eucalyptus plantations. In addition, studies can be carried out on specific birds using eucalyptus plantation particularly on their habitat use and foraging strategies. This is to reveal in-depth understanding on eucalyptus plantation suitability for local birds in Afrotropical montane.

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Appendices

Appendix 1: A notice of mode of payment for collecting/harvesting forest related resources in one of the forest stations in the study area (note the resources: Fuelwood, Grazing, Grass/Withies/Moss/Asparagus. Seedlings and cuttings etc).

PAYBILL Number ---770240

CHUKA FOREST STATION-THARAKA NITHI

KFS REVENUE SOURCES

CODE--- A/C	SOURCES
110201	Timber
110202	Mangrove Activities
120101	Professional Fee
120102	Tuition
120202	Water Charges
120203	Aircraft
130101	Movement Permits
130102	Monthly Fuel License
130104	Saw millersLicense
130105	Poles
130106	Fuel wood
130107	Quarrying/ Sand /Soil
130109	Ecotourism Licenses
130111	Grazing Fees
130113	Recreation Activities
130115	Grass /Withies/ Moss /Asparagus
130116	Hire Of Plant And Equipment
130201	Gross Tariff
130202	Examination Fees
130203	Students Id
130204	Accommodation
140101	House Rent
140201	Leases
140202	Land Rent RENT
140203 ✓	Pelis /Shamba Rent
140301	Bonded Items
150101	Seedlings &Cuttings

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Appendix 2: Study variable definitions, how measured and relevance to the study

Study variable	Definition	Units	Measurement method	Rationale for inclusion	Chapter
			1) Bird (i.e. dependent) variables		
Species richness (α-diversity) and abundance	Number of species within a given area and number of individual birds.	Count	50 m radius point counts. Overall species richness and abundance also broken down by study forest type and by functional groupings (e.g FD, DG, KMEB, IUCN red list, ATHB)	Species richness and abundance is a basic objective of field studies in community ecology (Boulinier et al, 1998).	Chapter 5 and 6.
Community composition (β-diversity)	Number of functional groups such as in FD, DG, KMEB, ATHB - differences and similarity (Magurran &McGill, 2011)	Proportions and Indices	Determining functional group species dissimilarity/similarity ANOSIM, SIMPER and NMDS illustration as measure of association between groups.	-Help to determine homogenizing or diversifying effects of disturbance/human actions (Magurran and McGill, 2011)	Chapter 6
			2) Habitat characteristic variables		
			(a) vegetation structure variables		
Tree height and tree DBH	Vascular plants ≥7.6 cm DBH and at least 5 m height (Height-vertical distance from the base to tip of the highest branch).	Tree heights (m), DBH (cm)	Tree height- using Nikon forestry laser rangefinder and visual estimation. DBH-using dbh meter (Diameter tape) at breast height level (1.3 m from the base) and following recommendation of Dahdouh-Guebas & Koedam (2006).	Tree heights and DBH are important habitat feature to measure in bird studies related to habitats (Bibby et al., 2000; Bibby et al., 2000; Peh et al., 2005 Thinh, 2006 Huang et al, 2014). DBH distributions can provide information on tree sizes (Ngoc Le et al., 2016).	Chapter 4, 5 and 6
Canopy height	The height of canopy at levels: low (≤ 10m), mid (11-20) and high (>20m).	Meters (m)	Estimated using Laser Rangefinder and visual estimation in dense vegetation within each 10 m radius point count (Lee and Marsden, 2008).	It is an *important* indicator of *forest* biomass, species diversity, site quality and ecosystem functioning (Tao et al, 2016).	
Canopy cover	The percentage of the ground covered by a vertical projection of the outermost perimeter of the natural spread of the foliage of plants	Percentage (%)	Use of GRS Densitometer. Taking the reading in five points i.e. center of point count, and at the four quadrants of each of 10 m radius to increase the reliability of the measurement.	Has been found to correlate with bird species richness and diversity by past researchers (e.g., MarcArthur 1961; Thinh, 2006).	
Vegetation cover at: 0-1m, 1-3m, 3-	Vegetation cover	Percentage (%); diversity	Visual estimation of percentage coverage of each height category	FHD correlate with species richness and diversity (MarcArthur 1961). Intensities of disturbances affect	

Study variable	Definition	Units	Measurement method	Rationale for inclusion	Chapter
5m, 5-8m, > 8m profile from forest floor.	(Volume of vegetation in percentage) from ground level to mid and upper levels. Will be used to generate Foliage Height diversity index using Shannon-weiner formula.	index (Shannon-Weiner, H').	within a radius of 10 m around the center of point count.	foliage profiles at different heights and there can be a negative correlation between the amount of vegetation in the upper and lower strata (Bibby et al., 2000).	
Litter depth	Depth of loose fallen vegetation especially twigs, leaves that covers the ground.	Centimeters (cm).	Using a measuring ruler to measure litter depth directly, averaged over 5 representative samples measured at the center of point count and at center of each 10 m quadrant.	Birds are exposed to it and its contents (microbes and moisture). Influence invertebrates' presence (Willson and Comet, 1996) hence distribution of birds, especially the understory (e.g., Banks et al., 2017).	
No. of Standing snag	Dead, decaying and standing trees (≥1 m) that can easily form cavities and loose barks.	Counts	Directly counting the number of dead standing trees within each of the 10m radius plot within the point count.	Easily used by birds to make nests (Vázquez and Renton, 2015), store food supplies and get food sources. Can be used as a measure of disturbance, human target it to harvest as firewood (Pers. Obser in Mount Kenya forest).	
			b) Disturbance variables		
No. of fallen/downed trees	All fallen/down lying felled trees, natural caused felled trees including Dead and downed rotting trees.	Counts	Counting the number of fallen trees/downed, either by natural causes or human/animals within the 10 m radius of the point count. Human caused are determined by looking at cuts, burns while animal caused by looking at evidence of animal pushes and other animal features (all classified as fallen trees)	Other plants can take root on downed logs (act as nurse log) hence can grow well on rotting trees and influence bird habitat. It can also be used as a measure of human disturbance affecting birds. Fallen/falling trees open up canopy influencing light penetration and undergrowth characteristics influencing birds (Pers. Observ).	
No. of cut trees	Any vascular woody vegetation from small trees, and larger trees that has been freshly cut and with signs of cutting from the roots, trunks, and branches.	Counts	Physically counting all the cut trees within the 10 m radius of the point count.	Forest cutting is a major anthropogenic disturbance (Shahabuddin and Kumar, 2006), i.e., associate with removal of habitat components that would have been available for bird use. Small-scale disturbance is a significant process in all major forest biomes (Forsman et al 2013).	

Study variable	Definition	Units	Measurement method	Rationale for inclusion	Chapter
No. of trails/paths	Human paths/trails intersecting the 10 m radius circle around the point count	Counts	Physically counting the paths intersecting the 10 m radius circle of the point counts	Presence of people in forests can disturb wildlife (perceived as potential predators) (Bötsch et al., 2018). Human activities rely on trails, which intersect an otherwise contiguous habitat (Bötsch et al., 2018). Number and width of paths may be useful in characterizing human disturbance (Bibby et al., 2000).	
c) Slope, altitude, and distance from forest edge					
Slope	Mean gradient from two measurements, one up and one down the general slope pattern, using a clinometer	degrees	Measured using clinometer expressed as a mean gradient over two points (10 m apart). Two 1 m rod was used, one placed 10 m up slope/downslope and the other in the centre of point count, using clinometer with the aid of the two rods. Center of point count will be used as reference point.	Determine bird distribution (Bibby et al. 2000) and forest vegetation structure (Pascal and Pelissier 1996).	Chapter 4, 5 and 6.
Altitude	The height of a point count above sea level	**Meters (m)**	Measured using hand-held Garmin GPSMAP 64S that automatically record the altitude in meters to accuracy of between 10 to 20 m.	It is a determinant of bird distribution (Bibby et al., 2000). It is powerful in identifying bird-habitat relationships and relatively easy and quick to collect (Lee and Marsden, 2008; Kim et al., 2018).	
Distance to the forest edge	Distance to human settlement /agricultural land from each point count.	Meters (m) and Kilometers (kms)	Using handheld GPS, the distance measured directly from each point count location to the edge/boundary or vice versa.	Important in many bird species, whether as an indicator of general forest quality (e.g., Gehlhausen et al. 2000), or a correlate of disturbance or hunting (Lee and Marsden, 2008).	
Forest types	Type of forest e.g., undisturbed natural, disturbed natural and eucalyptus forest.	N/A	Determined through pilot study, disturbance level in indigenous forest determined using number of cut trees, paths/trails	Forests hold 50% of species of conservation importance (Holbech, 2009).	

FD = forest dependence; FG= dietary guilds; KMEB = Kenyan mountains endemic birds; IUCN red list = International Union for conservation of Nature red listing, ATHB=Afrotropical highlands biome restricted range specis; DBH = diameter at breast height; FHD = foliage height diversity; ANOSIM = analysis of similarities; SIMPER = Percentage similarities, NMDS = non-metric multidimensional scaling.

Appendix 3: (a) Controlling for seasonal differences among habitat characteristics, species richness and abundance, and species compositions in the study area.

For chapter 4, to control for seasons, bivariate analysis was done between key response variables that significantly differed across forest types, and that had also the highest means in each forest types as shown in Table 4.2, and seasons. For undisturbed forest, seasons did not strongly predict the following: % vegetation covers at 5-8 m, > 8 m, canopy cover, and litter depth respectively (i.e., F = 8.266, df = 3, p = 0.031; F = 3.699, df = 3, p = 0.012; F = 11.713, df = 3, p = 0.025; F = 12.845, df = 3, p = 0.161). Seasons also did not significantly predict the number of snags (i.e., χ^2 Likelihood ratio = 0.119, df = 3, p = 0.989). For disturbed forest type, seasons did not significantly predict number of fallen trees and number of saplings respectively (i.e., GLM, χ^2 Likelihood ratio = 0.705, df = 3, p = 0.872; χ^2 Likelihood ratio = 0.141, df = 3, p = 0.986) and did not strongly predict % vegetation cover at 3-5 m and FHD (i.e., F =7.397, df = 3, p = 0.0371; F = 9.144, df = 3, p = 0.049). Seasons did not significantly predicted vegetation covers at 0-1 m, number of cut trees and trails across eucalyptus plantation forest type (i.e., F = 2.325, df = 3, p = 0.077; χ^2 Likelihood ratio = 0.333, df = 3, p = 0.954; χ^2 Likelihood ratio = 0.119, df = 3, p = 0.989).

For chapter 5, to control for seasons, a bivariate analysis was done between seasons and species richness and relative abundance across each forest type. In all cases, seasons did not strongly predict the species richness and relative abundance and hence seasonality was not considered further in this study. For example, in disturbed forest; species richness χ^2 Likelihood ratio = 12.301, df = 3, p = 0.031; relative abundance χ^2 Likelihood ratio = 24.404, df = 3, p = 0.059. In undisturbed forest, species richness χ^2 Likelihood ratio = 24.676, df = 3, p = 0.061; relative abundance χ^2 Likelihood ratio = 16.790, df = 3, p = 0.024. Eucalyptus plantation, species richness χ^2 Likelihood ratio = 16.790, df = 3, p = 0.039; relative abundance χ^2 Likelihood ratio = 5.962, df = 3, p = 0.059.

For Chapter 6, to control for seasons, a measure of similarity was done using ANOSIM and illustrated using NMDS as shown in Appendix 3 a, and 3 b. ANOSIM showed that there is less dissimilarity of the species composition between seasons (R = 0.169) (i.e., since the closer the value of Global *R* is to 1, the more dissimilarity the species

composition are between the seasons. This is also illustrated through NMDS in Appendix 3 b below, where there is a greater overlap and clustering of species compositions across seasons.

Appendix 3: (b) NMDS ordination for overall birds' species and abundance composition across seasons (Long rainy season (forest green with dots), Short dry season (coral with plus sign), Short rainy season (light green with squares), and Long dry season (brown with x sign). The borderline delineates bird species composition for each season.
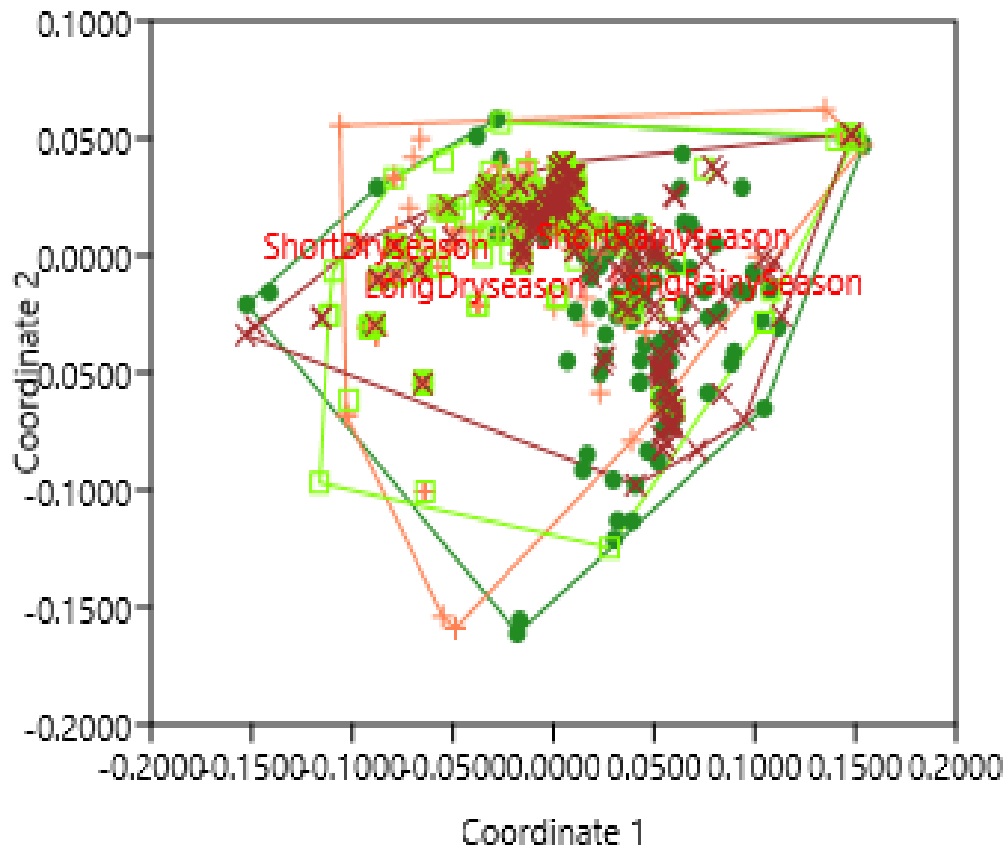
Appendix 4: Descriptive statistics of habitat characteristic variables across forest types (study site data pooled)

Habitat characteristic variable	Forest types	N	Mean	SE Mean	Minimum	Maximum	Range
			Vegetation variables				
Tree height (m)	Undisturbed	75	14.73	0.59	2.25	34.75	32.5
	Disturbed	75	13.16	0.45	1.5	21.5	20
	Plantation	40	17.45	0.94	5	27.5	22.5
Tree DBH (cm)	Undisturbed	75	91.52	4.81	30.38	224	193.62
	Disturbed	75	63.46	3.48	6.5	172.13	165.63
	Plantation	40	68.31	4.04	21.88	115.5	93.62
Basal area (m²)	Undisturbed	75	0.79	0.08	0.07	3.94	3.87
	Disturbed	75	0.39	0.04	0	2.33	2.33
	Plantation	40	0.42	0.04	0.04	1.05	1.01
Tree distance from centre of point counts	Undisturbed	75	4.00	0.14	1.88	6.75	4.87
	Disturbed	75	3.10	0.13	0.5	6.13	5.63
	Plantation	40	4.38	0.22	1.38	7.25	5.87
% vegetation cover at 0-1 m	Undisturbed	75	40.44	2.8	5	90	85
	Disturbed	75	44.91	2.08	9.5	90	80.5
	Plantation	40	60.8	4.01	5	95	90
% vegetation cover at 1-3 m	Undisturbed	75	33.68	2.18	5	73.75	68.75
	Disturbed	75	39.44	1.99	9.75	80	70.25
	Plantation	40	37.55	3.39	4.25	77.5	73.25
% vegetation cover at 3-5 m	Undisturbed	75	29.85	1.56	7.5	63.75	56.25
	Disturbed	75	34.71	1.35	15	62.5	47.5
	Plantation	40	20.02	1.84	2.5	41.25	38.75
% vegetation cover at 5-8 m	Undisturbed	75	31.94	1.25	5	55	50
	Disturbed	75	29.64	0.9	16.25	52.5	36.25
	Plantation	40	15.34	1.15	2.5	28.75	26.25
% vegetation cover at > 8 m	Undisturbed	75	26.05	1.44	1.25	60	58.75
	Disturbed	75	21.15	0.86	8.75	43.75	35
	Plantation	40	12.02	0.87	0	25.75	25.75
% Canopy cover	Undisturbed	75	80.22	0.68	58.45	91.55	33.1
	Disturbed	75	71.57	2.01	6.03	88.7	82.67
	Plantation	40	46.79	3.05	0	68.55	68.55
Foliage height diversity FHD	Undisturbed	75	1.45	0.01	0.95	1.57	0.62
	Disturbed	75	1.49	0.01	1.38	1.57	0.19
	Plantation	40	1.34	0.03	0.71	1.57	0.87
Litter depth (cm)	Undisturbed	75	2.48	0.09	1.35	5.03	3.68
	Disturbed	75	2.23	0.07	1.08	4.03	2.95
	Plantation	40	2.09	0.10	1.08	3.38	2.3
Number of saplings	Undisturbed	75	6.63	0.60	0	20	20
	Disturbed	75	8.85	0.47	1	20	19
	Plantation	40	0.38	0.12	0	3	3
Number of snags	Undisturbed	75	1	0.09	0	4	4
	Disturbed	75	0.73	0.09	0	3	3

Habitat characteristic variable	Forest types	N	Mean	SE Mean	Minimum	Maximum	Range
	Plantation	40	0	0	0	0	0
	Undisturbed	75	7.25	0.12	5	10	5
	Disturbed	75	7.14	0.16	4	10	6
Low-canopy height 0-10 m	Plantation	40	7.23	0.25	2	10	8
	Undisturbed	75	13.61	0.39	0	20	20
	Disturbed	75	14.11	0.24	11	20	9
Mid-canopy height 11-20 m	Plantation	40	13.73	0.71	0	19	19
	Undisturbed	75	18.64	1.12	0	33	33
	Disturbed	75	20.72	0.87	0	30	30
High-canopy height > 20 m	Plantation	40	18.32	1.61	0	28	28
Disturbance variables							
	Undisturbed	75	0.47	0.10	0	4	4
	Disturbed	75	1.89	0.15	0	7	7
Number of cut trees	Plantation	40	2.08	0.18	0	5	5
	Undisturbed	75	0.67	0.09	0	3	3
	Disturbed	75	1.61	0.11	0	4	4
Number of trails	Plantation	40	1.88	0.19	0	4	4
	Undisturbed	75	1.27	0.11	0	4	4
	Disturbed	75	1.32	0.10	0	3	3
Number of fallen trees	Plantation	40	0.3	0.08	0	2	2
Environmental variables							
	Undisturbed	75	10.68	0.852	0.75	30.75	30
	Disturbed	75	11.08	0.917	1.5	29.5	28
Slope	Plantation	40	7.63	0.717	1.25	19.3	18.05
	Undisturbed	75	2098	38	1666	2576	910
	Disturbed	75	1915.9	27.5	1558	2255	697
Altitude	Plantation	40	2032.6	47.4	1552	2299	747
Distance of point count from edge	Undisturbed	75	3767.1	41.8	3010	4410	1400
	Disturbed	75	710.2	37.4	200	1285	1085
	Plantation	40	739.4	64.2	200	1420	1220

Appendix 5: (a) Results of VIF calculation to check for multicollinearity among habitat characteristic variables

Coefficients[a]	Collinearity Statistics		Coefficients[a]	Collinearity Statistics		Coefficients[a]	Collinearity Statistics	
Model	Tolerance	VIF	Model	Tolerance	VIF	Model	Tolerance	VIF
Tree DBH	0.069	**14.539**	Basal area	0.610	1.639	Tree distance from centre of point count	0.706	1.416
Basal area	0.076	**13.222**	Tree distance from centre of point count	0.718	1.393	% Veg. cover at 0-1 m	0.234	4.282
Tree distance from centre of point count	0.696	1.437	% veg. cover at 0-1 m	0.235	4.252	% veg. cover at 1-3 m	0.175	5.700
% veg. cover 0-1 m	0.231	4.333	% veg. cover at 1-3 m	0.175	5.705	% veg. cover at 3-5 m	0.257	3.891
% veg. cover at 1-3 m	0.176	5.688	% veg. cover at 3-5 m	0.257	3.890	% veg. cover at 5-8 m	0.246	4.059
% veg. cover at 3-5 m	0.258	3.881	% veg. cover at 5-8 m	0.246	4.059	% veg. cover at > 8	0.350	2.855
% veg. cover at 5-8 m	0.248	4.026	% veg. cover at > 8 m	0.350	2.855	% canopy cover	0.466	2.147
% veg. cover at > 8 m	0.351	2.847	% Canopy cover	0.474	2.111	Canopy hgt at 0-1 m	0.826	1.211
% Canopy cover	0.465	2.152	Canopy hgt at 0-10 m	0.824	1.214	Canopy hgt at 11-20 m	0.655	1.526
Canopy hgt at 0-10m	.8205	1.212	Canopy hgt at 11-20 m	0.653	1.532	Canopy hgt at > 20	0.607	1.649
Canopy hgt at 11-20 m	0.653	1.531	Canopy hgt at > 20 m	0.607	1.648	FHD	0.350	2.861
Canopy hgt at > 20 m	0.612	1.635	FHD	0.348	2.870	Number of saplings	0.538	1.860
FHD	0.347	2.881	Number of saplings	0.538	1.860	Litter depth	0.544	1.839
Number of saplings	0.553	1.809	Litter depth	0.548	1.825	Number of cut trees	0.507	1.974
Litter depth	0.544	1.840	Number of cut trees	0.508	1.968	Number of trails	0.543	1.840
Number of cut trees	0.532	1.880	Number of trails	0.546	1.831	Number of snags	0.564	1.775
Number of trails	0.573	1.747	Number of snags	0.565	1.770	Number of fallen trees	0.578	1.731
Number of snags	0.562	1.778	Number of fallen trees	0.579	1.726	Slope	0.725	1.379
Number of fallen trees	0.579	1.726	Slope	0.723	1.383	Altitude	0.489	2.047
Slope	0.723	1.384	Altitude	0.495	2.020	Distance from forest boundary	0.361	2.772
Altitude	0.491	2.036	Distance from forest boundary	0.361	2.771	Tree heights	0.578	1.731
Distance from forest boundary	0.363	2.752	Tree height	0.631	1.585	Tree DBH	0.508	1.968

Appendix 5: (b) Pearson Correlation of habitat characteristic variables

Row/column variables (top to bottom): Dist to boundary, Altitude, Slope, # fallen trees, # snags, # trails, # cut trees, # sapling, L. Depth, FHD, C. hgt > 20 m, C. hgt, C.hgt, % Veg Cover, % Veg Cover, % Veg Cover, % Veg Cover, % Canopy Cover, % Veg Cover, % Veg Cover, Tree dist to point, Tree Basal, Dbh, Tree.

Visible lower‑triangular correlation block (last variables):

	Tree Height	Tree Dbh	Basal Area	Tree dist to point count	% Veg Cover	% Veg Cover	% Veg Cover	% Veg Cover 5-8 m	% Veg Cover >8 m	% Canopy Cover Canopy height	% Canopy Cover 0-10 m
Tree Height	1										
Tree Dbh	.484**	1									
Basal Area	.388**	.956**	1								
Tree dist to point count	.329**	.321**	.264**	1							
% Veg Cover	.178*	.130	.110	.118	1						
% Veg Cover	.152*	.125	.112	.010	.696**	1					
% Veg Cover	-.073	.000	-.002	-.230**	.082	.589**	1				
% Veg Cover 5-8 m	-.198**	.001	-.012	-.178*	-.438**	-.198**	.361**	1			
% Veg Cover >8 m	-.013	.176*	.153*	-.022	-.446**	-.362**	-.056	.664**	1		
% Canopy Cover Canopy height	.138	.333**	.271**	.036	-.230*	.048	.316**	.488**	.435**	1	
% Canopy Cover 0-10 m	.042	-.058	-.091	.001	.000	-.039	.031	.160*	.134	.143*	1

Correlation matrix (values read column-wise under each variable heading; * p < 0.05, ** p < 0.01)

C. height at 11-20 m	Canopy height at > 20 m	FHD	# sapling	L. Depth	# cut trees	# trails	# snags	# fallen trees	Slope	Altitude	Dist to boundary
.092	.139	-.033	-.228**	-.077	.109	-.040	-.126	-.134	-.122	.107	-.014
.144*	.095	.006	-.098	.075	-.194**	-.130	.003	-.018	-.111	.252**	.333**
.141	.055	-.019	-.109	.049	-.201**	-.134	.007	-.035	-.126	.259**	.327**
-.072	.040	-.207**	-.269**	-.052	-.093	.040	-.099	-.199**	-.142	.111	.166*
-.137	-.037	-.359**	-.324**	-.307**	.019	.183*	-.325**	-.145*	-.252**	.167*	-.228**
-.035	.083	.085	.010	-.338**	-.011	.108	-.181*	.089	-.298**	-.023	-.152*
-.022	.131	.520**	.354**	-.143*	-.037	.017	.002	.218**	-.169*	-.178*	-.006
-.032	.141	.537**	.379**	.411**	-.178*	-.158*	.324**	.192**	.261**	-.314**	.312**
.108	.207**	.321**	.228**	.502**	-.222**	-.253**	.339**	.108	.390**	-.231**	.366**
.058	.306**	.378**	.341**	.248**	-.255**	-.252**	.277**	.284**	.094	-.134	.428**
.180*	.235**	.156*	.065	.033	.083	.019	-.022	.014	.090	-.104	.007
1	.311**	.312**	.098	.156*	-.016	.052	.052	.144*	.001	-.020	-.030
	1	.149*	.200**	.031	-.060	.143*	.143*	.083	.020	-.347**	-.076
		.232**	.075	.092	.092	.096	.096	.203**	.111	-.145*	.013
		.420**	.020	-.041	-.175*	-.204**	.356**	.104	-.300**	-.236**	-.557**
		1	1	-.177*	-.165*	-.380**	.466**	.261**	-.116	-.116	-.495**
				1	.490**	-.251**	-.064	-.125	.000	.346**	.346**
					1	.448**	-.251**	-.189**	.056	.158*	.158*
						1	.185*	-.300**	-.212**	-.054	.056
							.068	-.116	-.159*	-.212**	
							1	.261**	-.054		
								1	1	1	1

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Appendix 6: (a) The vegetation structure predictor variables that improved the Poisson regression model (p < 0.05) and those that did not (p > 0.05) with response variables. The one with p < 0.05 shows predictors representing a significant improvement of the model over a null model (i.e., with no predictors) and thus represent good fit for the response variables.

Response variables	Likelihood ratio χ^2	df	p-value
Overall species richness	13.739	16	0.686
Overall species abundance	1.078	16	1.000
FS richness	48.485	16	0.000
FS abundance	48.166	16	0.000
FG richness	13.768	16	0.683
FG abundance	2.398	16	1.000
FV richness	36.662	16	0.002
FV abundance	61.708	16	0.000
NF richness	111.388	16	0.000
NF abundance	189.809	16	0.000
Carnivore richness	20.709	16	0.190
Carnivore abundance	27.105	16	0.040
Frugivore richness	145,302	16	0.000
Frugivore abundance	29.738	16	0.019
Granivore richness	43.638	16	0.000
Granivore abundance	49.843	16	0.000
Insectivore richness	32.618	16	0.008
Insectivore abundance	3.575	16	0.999
Nectarivore richness	27.113	16	0.040
Nectarivore abundance	44.624	16	0.000
Omnivore richness	116.613	16	0.000
Omnivore abundance	111.945	16	0.000
ATHB richness	24.497	16	0.079
ATHB abundance	5.097	16	0.995

Appendix 6: (b) The disturbance predictor variables that improved the Poisson regression model (p < 0.05) and those that did not (p > 0.05) with response variables. The one with p < 0.05 shows predictors representing a significant improvement of the model over a null model (i.e., with no predictors) and thus represent good fit for the response variables.

Response variables	Likelihood ratio χ^2	df	p-value
Overall species richness	6.720	3	0.081
Overall species abundance	20.996	3	0.000
FS richness	13.045	3	0.005
FS abundance	10.392	3	0.016
FG richness	4.041	3	0.257
FG abundance	0.672	3	0.880
FV richness	20.913	3	0.000
FV abundance	28.151	3	0.000
NF richness	45.050	3	0.000
NF abundance	80.630	3	0.000
Carnivore richness	1.149	3	0.765
Carnivore abundance	0.714	3	0.870
Frugivore richness	41.039	3	0.000
Frugivore abundance	7.931	3	0.047
Granivore richness	12.189	3	0.007
Granivore abundance	18.337	3	0.000
Insectivore richness	7.015	3	0.071
Insectivore abundance	0.697	3	0.874
Nectarivore richness	3.535	3	0.316
Nectarivore abundance	8.787	3	0.032
Omnivore richness	35.508	3	0.000
Omnivore abundance	42.164	3	0.000
ATHB richness	3.779	3	0.286
ATHB abundance	0.396	3	0.941

**Appendix 6: (c)** The slope, altitude and distance from forest edge predictor variables that improved the **Poisson** regression model (p < 0.05) and those that did not (p > 0.05) with response variables. The one with p < 0.05 shows predictors representing a significant improvement of the model over a null model (i.e., with no predictors) and thus represent good fit for the response variables.

Response variables	Likelihood ratio χ^2	df	p-value
Overall species richness	11.122	3	0.011
Overall species abundance	0.586	3	0.900
FS richness	14.436	3	0.002
FS abundance	11.703	3	0.008
FG richness	0.593	3	0.898
FG abundance	0.844	3	0.839
FV richness	36.087	3	0.000
FV abundance	43.951	3	0.000
NF richness	64.477	3	0.000
NF abundance	102.130	3	0.000
Carnivore richness	1.939	3	0.585
Carnivore abundance	2.820	3	0.420
Frugivore richness	89.534	3	0.000
Frugivore abundance	17.522	3	0.001
Granivore richness	38.330	3	0.000
Granivore abundance	167.484	3	0.000
Insectivore richness	7.318	3	0.062
Insectivore abundance	2.750	3	0.432
Nectarivore richness	6.509	3	0.089
Nectarivore abundance	11.350	3	0.010
Omnivore richness	27.726	3	0.000
Omnivore abundance	37.485	3	0.000
ATHB richness	18.665	3	0.000
ATHB abundance	4.832	3	0.185

Appendix 7: a) (i) Normality of residuals of regression of vegetation structure variables as predictors with bird diversity indices as response variable (ii) normal P-P plot of regression standardised residuals using vegetation variables as predictors and bird diversity indices as response variable.
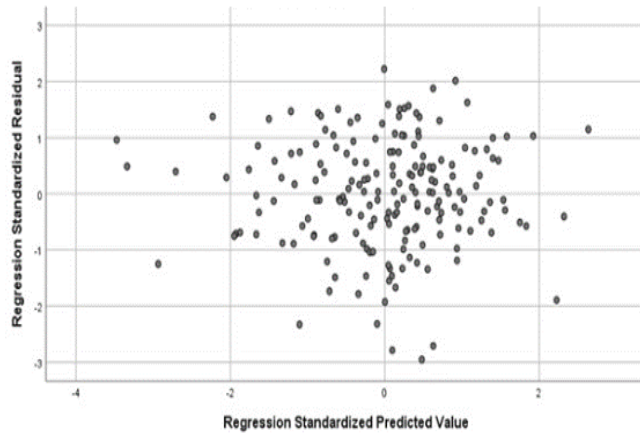
(i)
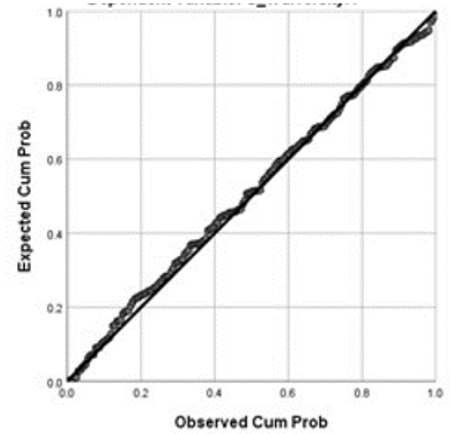
(ii)

Appendix 7: b) (i) Normality of residuals of regression of human disturbance variables as predictors with bird diversity indices as response variable (ii) normal P-P plot of regression standardised residuals using disturbance variables as predictors and bird diversity indices as response variable.

(i)

(ii)

Appendix 7: c) (i) Normality of residuals of regression of slope, altitude and distance from forest edge as predictors with bird diversity indices as response variable (ii) normal P-P plot of regression standardised residuals using environmental variables as predictors and bird diversity indices as response variable.
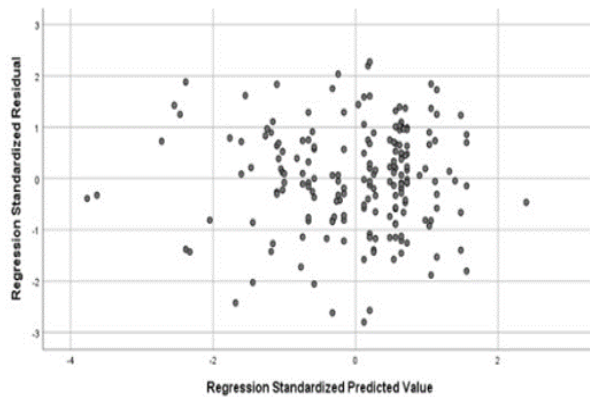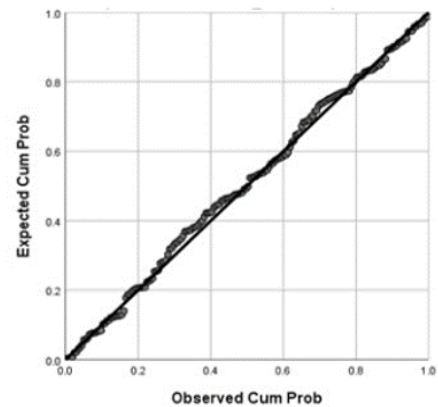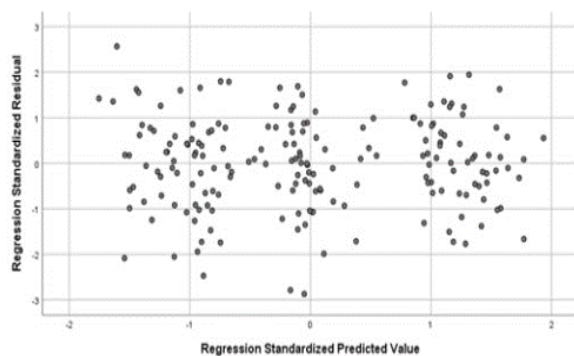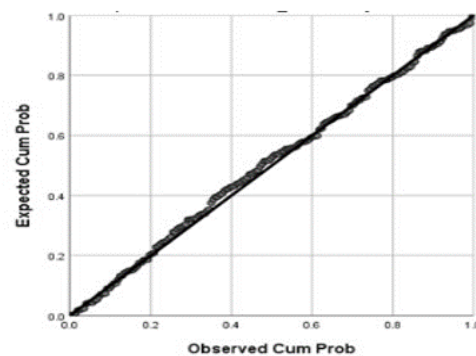

(i)   (ii)

Appendix 8: Overall bird species list. FD = forest dependency: FS = forest specialists, FG= forest generalists, FV = forest visitors, NF = non-forests. DG = dietary guilds, IUCN Cat = IUCN Red list categorisation: VU = vulnerable, NT = near threatened, LC = least concern. ATHB = afrotropical highlands biome. Dist.= disturbed forest, Plant. = plantation forest, Undist. = undisturbed forest. Unique species to undisturbed forest = ▬, unique species to disturbed forest = ▬, unique species to plantation forest = ▬, shared species across all forest types = Ψ, shared species between disturbed and undisturbed forest types = Ω, shared species between disturbed and plantation forest type = Φ, shared species between undisturbed and plantation forest types = ώ.

Species	Scientific name	Family	FD	DG	IUCN Cat	ATHB	Undist.	Dist.	Plant.
Abbott's Starling Ω	*Poeoptera femoralis*	Sturnidae	FS	Frugivore	VU	✓	✓	✓	
Abyssinian Crimsonwing	*Cryptospiza salvadorii*	Estrildidae	FG	Granivore	LC	✓			✓
Abyssinian Ground Thrush Ω	*Zoothera piaggiae*	Turdidae	FS	Insectivore	LC	✓	✓	✓	
African Black Duck	*Anas sparsa*	Anatidae	NF	Omnivore	LC			✓	
African Broadbill Ω	*Smithornis capensis*	Calyptomenidae	FS	Insectivore	LC		✓	✓	
African Citril	*Serinus citrinelloides*	Fringillidae	FV	Granivore	LC				✓
African Crowned Eagle Ω	*Stephanoaetus coronatus*	Accipitridae	FS	Carnivore	NT		✓	✓	
African Cuckoo	*Cuculus gularis*	Cuculidae	NF	Insectivore	LC				✓
African Dusky Flycatcher Ψ	*Muscicapa adusta*	Muscicapidae	FG	Insectivore	LC		✓	✓	✓
African Emerald Cuckoo Ψ	*Chrysococcyx cupreus*	Cuculidae	FG	Insectivore	LC		✓	✓	✓
African Firefinch Φ	*Lagonosticta rubricata hildebrandti*	Estrildidae	NF	Granivore	LC			✓	✓
African Fish Eagle	*Haliaeetus vocifer*	Accipitridae	NF	Carnivore	LC			✓	
African Goshawk	*Accipiter tachiro*	Accipitridae	FG	Carnivore	LC			✓	
African Green Pigeon Ω	*Treron calvus*	Columbidae	FG	Frugivore	LC		✓	✓	
African Harrier-hawk	*Polyboroides typus*	Accipitridae	FV	Carnivore	LC				✓
African Hill Babbler Ω	*Sylvia abyssinica*	Sylviidae	FS	Insectivore	LC	✓	✓	✓	
African Paradise Flycatcher Ψ	*Terpsiphone viridis*	Monarchidae	FV	Insectivore	LC		✓	✓	✓
African Wood Owl Ω	*Strix woodfordii*	Strigidae	FG	Insectivore	LC		✓	✓	
Amethyst Sunbird Φ	*Chalcomitra amethystina*	Nectariniidae	FV	Nectarivore	LC			✓	✓
Augur Buzzard	*Buteo augur*	Accipitridae	NF	Carnivore	LC				✓
Ayres's Hawk Eagle	*Hieraaetus ayresii*	Accipitridae	FG	Carnivore	LC			✓	
Baglafecht Weaver	*Ploceus baglafecht*	Ploceidae	FV	Omnivore	LC	✓			✓
Bar-tailed Trogon Ω	*Apaloderma vittatum*	Trogonidae	FS	Insectivore	LC	✓	✓	✓	

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Common Name	Scientific Name	Family		Diet	Status				
Black Cuckoo-shrike	*Campephaga flava*	Campephagidae	FV	Insectivore	LC			✓	
Black Saw-wing	*Psalidoprocne holomelas*	Hirundinidae	FV	Insectivore	LC		✓		
Black-and-white Mannikin	*Spermestes bicolor*	Estrildidae	FV	Granivore	LC				✓
Black-backed Puffback Ψ	*Dryoscopus cubla*	Malaconotidae	FG	Insectivore	LC		✓	✓	✓
Blackcap	*Sylvia atricapilla*	Sylviidae	FG	Insectivore	LC				✓
Black-collared Apalis Φ	*Oreolais pulcher*	Cisticolidae	FG	Insectivore	LC	✓		✓	✓
Black-fronted Bush-Shrike Ω	*Malaconotus nigrifrons*	Malaconotidae	FS	Insectivore	LC		✓	✓	
Black-headed Apalis Ω	*Apalis melanocephala*	Cisticolidae	FS	Insectivore	LC		✓	✓	
Black-headed Waxbill Φ	*Estrilda atricapilla graueri*	Estrildidae	FG	Granivore	LC			✓	✓
Black-headed Weaver	*Ploceus melanocephalus*	Ploceidae	NF	Granivore	LC			✓	
Black-throated Apalis Ω	*Apalis jacksoni*	Cisticolidae	FS	Insectivore	LC		✓	✓	
Blue-headed Coucal	*Centropus monachus*	Cuculidae	NF	Carnivore	LC			✓	
Brimstone Canary	*Crithagra sulphurata sharpii*	Fringillidae	NF	Granivore	LC				✓
Bronze Mannikin	*Spermestes cucullata*	Estrildidae	NF	Granivore	LC				✓
Bronze Sunbird Φ	*Nectarinia kilimensis*	Nectariniidae	FV	Nectarivore	LC	✓		✓	✓
Brown Woodland Warbler Ψ	*Phylloscopus umbrovirens*	Phylloscopidae	FG	Insectivore	LC	✓	✓	✓	✓
Brown-backed Scrub Robin	*Cercotrichas hartlaubi*	Muscicapidae	FV	Insectivore	LC				✓
Brown-capped Weaver Φ	*Ploceus insignis*	Ploceidae	FS	Insectivore	LC	✓		✓	✓
Brown-chested Alethe Ω	*Alethe poliocephala*	Muscicapidae	FS	Insectivore	LC		✓	✓	
Brown-crowned Tchagra	*Tchagra australis*	Malaconotidae	NF	Insectivore	LC			✓	
Cabanis's Greenbul Ψ	*Phyllastrephus cabanisi*	Pycnonotidae	FS	Insectivore	LC		✓	✓	✓
Cape Robin Chat Φ	*Cossypha caffra iolaema*	Muscicapidae	FV	Insectivore	LC			✓	✓
Cardinal Woodpecker	*Dendropicos fuscescens*	Picidae	FV	Insectivore	LC				✓
Chestnut-throated Apalis Ω	*Apalis porphyrolaema*	Cisticolidae	FG	Insectivore	LC	✓	✓	✓	
Chiffchaff Ω	*Phylloscopus collybita abietinus*	Phylloscopidae	FG	Insectivore	LC		✓	✓	
Chin-spot Batis Φ	*Batis molitor*	Platysteiridae	NF	Insectivore	LC			✓	✓
Cinnamon Bracken WarblerΨ	*Bradypterus cinnamomeus*	Locustellidae	FG	Insectivore	LC	✓	✓	✓	✓
Cinnamon-chested Bee-eater Ψ	*Merops oreobates*	Meropidae	FG	Insectivore	LC	✓	✓	✓	✓
Collared Sunbird Ψ	*Hedydipna collaris*	Nectariniidae	FG	Nectarivore	LC		✓	✓	✓
Common Bulbul	*Pycnonotus barbatus*	Pycnonotidae	FV	Omnivore	LC				✓
Common Drongo Φ	*Dicrurus adsimilis*	Dicruridae	NF	Insectivore	LC			✓	✓
Common Fiscal	*Lanius collaris*	Laniidae	NF	Insectivore	LC				✓

Common name	Scientific name	Family		Diet	Status				
Common Stonechat	*Saxicola torquatus*	Muscicapidae	NF	Insectivore	LC				✓
Crested Guineafowl Ψ	*Guttera pucherani*	Numididae	FG	Granivore	LC		✓	✓	✓
Crowned Hornbill Ψ	*Lophoceros alboterminatus*	Bucerotidae	FV	Frugivore	LC		✓	✓	✓
Dark-capped Yellow Warbler Ω	*Chloropeta natalensis massaica*	Acrocephalidae	FG	Insectivore	LC		✓	✓	
Doherty's bush-shrike Ω	*Telophorus dohertyi*	Malaconotidae	FG	Insectivore	LC	✓	✓	✓	
Eastern Bronze-naped Pigeon Ω	*Columba delegorguei sharpei*	Columbidae	FS	Frugivore	LC		✓	✓	
Eastern Double-collared Sunbird Ψ	*Cinnyris mediocris*	Nectariniidae	FG	Nectarivore	LC	✓	✓	✓	✓
Eastern Honeybird	*Prodotiscus zambesiae ellenbecki*	Indicatoridae	FV	Insectivore	LC				✓
Eurasian Bee-eater Ψ	*Merops apiaster*	Meropidae	FV	Insectivore	LC		✓	✓	✓
Eurasian Nightjar	*Caprimulgus europaeus*	Caprimulgidae	NF	Insectivore	LC				✓
Evergreen Forest Warbler Ω	*Bradypterus lopezi*	Locustellidae	FS	Insectivore	LC		✓	✓	
Fine-banded Woodpecker Ω	*Campethera taeniolaema*	Picidae	FS	Insectivore	LC	✓	✓	✓	
Giant Kingfisher	*Megaceryle maxima*	Alcedinidae	NF	Carnivore	LC			✓	
Golden-breasted Bunting	*Emberiza flaviventris*	Emberizidae	NF	Granivore	LC				✓
Golden-winged Sunbird	*Nectarinia reichenowi*	Nectariniidae	FV	Nectarivore	LC	✓			✓
Great Sparrowhawk Ω	*Accipiter melanoleucus*	Accipitridae	FG	Carnivore	LC		✓	✓	
Greater Honeyguide	*Indicator indicator*	Indicatoridae	FV	Insectivore	LC			✓	
Green-headed Sunbird Ω	*Nectarinia verticalis viridisplendens*	Nectariniidae	FG	Nectarivore	LC		✓	✓	
Grey Apalis Ω	*Apalis cinerea*	Cisticolidae	FS	Insectivore	LC		✓	✓	
Grey Cuckoo-shrike Ψ	*Coracina caesia pura*	Campephagidae	FS	Insectivore	LC	✓	✓	✓	✓
Grey wagtail	*Motacilla cinerea*	Motacillidae	FG	Insectivore	LC				✓
Grey-backed Camaroptera Ψ	*Camaroptera brachyura*	Cisticolidae	FV	Insectivore	LC		✓	✓	✓
Grey-capped Warbler	*Eminia Lepida*	Cisticolidae	FV	Insectivore	LC				✓
Grey-headed Negrofinch Ψ	*Nigrita canicapilla*	Estrildidae	FG	Insectivore	LC		✓	✓	✓
Grey-headed Sparrow	*Passer griseus*	Passeridae	NF	Granivore	LC				✓
Hartlaub's Turaco Ψ	*Tauraco hartlaubi*	Musophagidae	FS	Frugivore	LC	✓	✓	✓	✓
Holub's Golden Weaver	*Ploceus xanthops*	Ploceidae	NF	Granivore	LC				✓
Hunter's Cisticola ẃ	*Cisticola hunteri*	Cisticolidae	FG	Insectivore	LC	✓	✓		✓
Jackobin's Cuckoo	*Oxylophus jacobinus*	Cuculidae	NF	Insectivore	LC				✓
Kenrick's Starling Ω	*Poeoptera kenricki*	Sturnidae	FS	Frugivore	LC	✓	✓	✓	
Klaas's Cuckoo Φ	*Chrysococcyx klaas*	Cuculidae	FV	Insectivore	LC			✓	✓
Lemon Dove Ω	*Aplopelia larvata*	Columbidae	FS	Frugivore	LC		✓	✓	

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Common Name	Scientific Name	Order/Family		Diet	Status				
Lesser Honeyguide Ω	*Indicator minor teitensis*	Piciformes	FV	Insectivore	LC	✓	✓		
Little Sparrowhawk	*Accipiter minullus*	Accipitridae	FV	Carnivore	LC				✓
Long-crested Eagle Φ	*Lophaetus occipitalis*	Accipitridae	FV	Carnivore	LC			✓	✓
Malachite Sunbird Ψ	*Nectarinia famosa cupreonitens*	Nectariniidae	FG	Nectarivore	LC		✓	✓	✓
Montane Nightjar	*Caprimulgus poliocephalus*	Caprimulgidae	FG	Insectivore	LC	✓			✓
Montane Oriole Ψ	*Oriolus percivali*	Oriolidae	FS	Frugivore	LC	✓	✓	✓	✓
Montane White-eye Ψ	Zosterops poliogaster	Zosteropidae	FG	Insectivore	LC	✓	✓	✓	✓
Mountain Buzzard	*Buteo oreophilus*	Accipitridae	FS	Carnivore	NT	✓	✓		
Mountain Greenbul Ψ	*Andropadus nigriceps*	Pycnonotidae	FS	Frugivore	LC		✓	✓	✓
Mountain Wagtail Φ	*Motacilla clara torrentium*	Motacillidae	FG	Insectivore	LC			✓	✓
Mountain Yellow Warbler	*Chloropeta similis*	Acrocephalidae	FG	Insectivore	LC	✓	✓		
Moustached Green Tinkerbird Ω	*Pogoniulus leucomystax*	Lybiidae	FS	Frugivore	LC	✓	✓	✓	
Northern Double-collared Sunbird Ψ	*Nectarinia preussi kikuyuensis*	Nectariniidae	FG	Nectarivore	LC	✓	✓	✓	✓
Olive Pigeon Ω	*Columba arquatrix*	Columbidae	FS	Frugivore	LC		✓	✓	
Olive Sunbird Ω	*Nectarinia olivacea*	Nectariniidae	FS	Nectarivore	LC		✓	✓	
Olive Thrush Ψ	*Turdus olivaceus*	Turdidae	FG	Insectivore	LC		✓	✓	✓
Orange Ground Thrush	*Zoothera gurneyi*	Turdidae	FS	Insectivore	LC	✓		✓	
Oriole-Finch	*Linurgus olivaceus*	Fringillidae	FG	Granivore	LC	✓		✓	
Plain martin	*Riparia paludicola ducis*	Hirundinidae	NF	Insectivore	LC				✓
Purple-throated Cuckoo-shrike	*Campephaga quiscalina martini*	Campephagidae	FS	Insectivore	LC			✓	
Red-backed Shrike ώ	*Lanius collurio*	Laniidae	NF	Insectivore	LC		✓		✓
Red-chested Cuckoo Φ	*Cuculus solitarius*	Cuculidae	FG	Insectivore	LC		✓	✓	
Red-eyed Dove Φ	*Streptopelia semitorquata*	Columbidae	FV	Granivore	LC		✓	✓	
Red-faced Cisticola ώ	*Cisticola erythrops sylvius*	Cisticolidae	FS	Insectivore	LC		✓		✓
Red-fronted Parrot Ω	*Poicephalus gulielmi massaicus*	Psittacidae	FS	Frugivore	LC		✓	✓	
Red-headed Bluebill	*Spermophaga ruficapilla*	Estrildidae	FG	Granivore	LC			✓	
Red-headed Weaver	*Anaplectes rubriceps*	Ploceidae	NF	Insectivore	LC				✓
Ring-necked Dove	*Streptopelia capicola*	Columbidae	FV	Granivore	LC				✓
Ruppell's Robin Chat Ψ	*Cossypha semirufa*	Muscicapidae	FG	Insectivore	LC	✓	✓	✓	✓
Scaly Francolin Ψ	*Francolinus squamatus*	Phasianidae	FG	Granivore	LC		✓	✓	✓
Scaly-throated Honeyguide	*Indicator variegatus*	Indicatoridae	FV	Insectivore	LC			✓	

Common Name	Scientific Name	Family		Diet	Status				
Shelley's Francolin Φ	*Francolinus shelleyi*	Phasianidae	NF	Granivore	LC			✓	✓
Silvery-cheeked Hornbill Ψ	*Bycanistes brevis*	Bucerotidae	FG	Frugivore	LC		✓	✓	✓
Singing Cisticola	*Cisticola cantans*	Cisticolidae	NF	Insectivore	LC				✓
Slender-billed Greenbul	*Andropadus gracilirostris*	Pycnonotidae	FS	Frugivore	LC		✓		
Slender-billed Starling	*Onychognathus tenuirostris theresae*	Sturnidae	FG	Omnivore	LC	✓	✓		
Speckled Mousebird	*Colius striatus*	Coliidae	NF	Frugivore	LC				✓
Spectacled Weaver	*Ploceus ocularis*	Ploceidae	FV	Insectivore	LC				✓
Streaky Seed-eater	*Serinus striolata*	Fringillidae	FV	Granivore	LC	✓			✓
Tacazze Sunbird Ψ	*Nectarinia tacazze*	Nectariniidae	FV	Nectarivore	LC	✓	✓	✓	✓
Tambourine Dove Ψ	*Turtur tympanistria*	Columbidae	FG	Granivore	LC		✓	✓	✓
Tawny-flanked Prinia Ψ	*Prinia subflava*	Cisticolidae	FV	Insectivore	LC		✓	✓	✓
Thick-billed Seed-eater Ψ	*Serinus burtoni*	Fringillidae	FS	Granivore	LC	✓	✓	✓	✓
Tree Pipit	*Anthus trivialis*	Motacillidae	FV	Insectivore	LC				✓
Tropical Boubou Ψ	*Laniarius aethiopicus*	Malaconotidae	FV	Insectivore	LC		✓	✓	✓
Variable Sunbird Φ	*Nectarinia venusta*	Nectariniidae	FV	Nectarivore	LC			✓	✓
Waller's Starling Ω	*Onychognathus walleri*	Sturnidae	FS	Frugivore	LC	✓	✓	✓	
White-bellied Tit Ψ	*Parus albiventris*	Paridae	FV	Insectivore	LC		✓	✓	✓
White-browed Coucal Φ	*Centropus superciliosus*	Cuculidae	NF	Carnivore	LC			✓	✓
White-browed Crombec	*Sylvietta leucophrys*	Macrosphenidae	FS	Insectivore	LC	✓		✓	
White-eared Barbet Ψ	*Stactolaema leucotis kilimensis*	Lybiidae	FG	Frugivore	LC		✓	✓	✓
White-eyed Slaty Flycatcher Ψ	*Melaenornis fischeri*	Muscicapidae	FG	Insectivore	LC	✓	✓	✓	✓
White-headed Wood-hoopoe Ω	*Phoeniculus bollei jacksoni*	Phoeniculidae	FS	Insectivore	LC		✓	✓	
White-starred Robin Ω	*Pogonocichla stellata*	Muscicapidae	FG	Insectivore	LC	✓	✓	✓	
White-tailed Crested Flycatcher Ω	*Trochocercus albonotata*	Stenostiridae	FS	Insectivore	LC	✓	✓	✓	
Willow Warbler Ψ	*Phylloscopus trochilus*	Phylloscopidae	FV	Insectivore	LC		✓	✓	✓
Yellow crowned canary	*Serinus canicollis flavivertex*	Fringillidae	FV	Granivore	LC				✓
Yellow Wagtail	*Motacilla flava*	Motacillidae	NF	Insectivore	LC				✓
Yellow-rumped Tinkerbird Ψ	*Pogoniulus bilineatus*	Lybiidae	FG	Frugivore	LC		✓	✓	✓
Yellow-whiskered Greenbul Ψ	*Andropadus latirostris*	Pycnonotidae	FG	Frugivore	LC		✓	✓	✓

Appendix 9: Glossary: Operationalisation of terms used in this study

Afromontane forests- forests found within Afromontane.

Afromontane- montane found in Africa continent and form part of Afrotropic subregion and its plant and animal species common to the mountains of Africa and the southern Arabian Peninsula.

Anthropogenic disturbance – human caused interruption of natural condition of ecosystem by either accessing for example forests and creating paths/trails, removal/harvesting of forest materials, cutting trees, putting hives, grazing livestock, and starting fires.

Community forest associations (CFAs) – it is a component of participatory forest management, and an element in forest conservation and management that is formed through coming together of communities that live adjacent to forests. The CFAs are assisted by forest management authorities to formally agree to use the forests sustainably to generate income and improve their livelihoods, and hence meant to ease pressure on the forest areas.

Disturbance- Events that cause change in the structure and composition of an ecosystem, and likely to impact the growth and survival of individual organisms in that ecosystem.

Disturbed forest – forest that is subject to events that cause change in the structure and composition of a forest ecosystem. In this study the disturbance events are mainly caused by anthropogenic activities.

Exotic plantations – plantations that consist of plants originating in a foreign country, not native. It may be used synonymously with the term 'introduced plantation'. In the current study it is represented by eucalyptus plantation which is part of the focus of the study.

Montane forests- the forest found within the montane and a target of the current study at its lower altitudes.

Montane- part of the tropical mountain that lies below the natural limit of trees (the tree line) and covered by forests.

Natural forest- it is a naturally growing forest. It is used synonymously with indigenous or native forest. It can be affected or not affected by human activities.

Participatory forest management - the active inclusion of rural communities in the management and utilisation of state-owned natural forests and woodlands.

Plantation forest- planted forests cultivated to form forest ecosystems established by planting or seeding in the process of afforestation and reforestation (e.g., Helms 1998). It consists of intensively managed, even aged, and regularly spaced stands of a single tree species, primarily for wood biomass production and ecological services (Zhang, et al., 2019).

Tropical montane cloud forests- a tropical evergreen, montane, moist forest characterised by a persistent, frequent, or seasonal low-level cloud cover, usually at the canopy level, and exhibit an abundance of mosses covering the ground and vegetation. It does not form a target of the current study.

Tropical mountains- Mountains that are within the world's tropical zone i.e., zones between tropic of Cancer and tropic of Capricorn.

Undisturbed forest – natural or native forest with no or little anthropogenic disturbance, hence relatively intact. It can be synonymous with primary forest.

Firewood/Fuelwood – wood used as fuel**.**