Afromontane avian assemblages and land use in the Bale Mountains of Ethiopia: patterns, processes and conservation implications

by

Addisu Asefa Mitiku

Submitted in partial fulfillment of the requirements for the degree

M.Sc. Zoology

In the Faculty of Natural and Agricultural Science

University of Pretoria

Pretoria

November 2013

Declaration

I, **Addisu Asefa Mitiku,** declare that the dissertation, which I hereby submit for the degree **M.Sc. Zoology** at the University of Pretoria, is my own work and has not previously been submitted by me for a degree at this or any other tertiary institution.

SIGNATURE:

DATE: November 2013

Afromontane avian assemblages and land use in the Bale Mountains of Ethiopia: patterns, processes and conservation implications

Student: Addisu Asefa Mitiku[1]

Supervisors: Prof. Andrew E. McKechnie[1], Prof. Berndt J. van Rensburg[1,2], Dr. Anouska A. Kinahan[3] and Dr. Andrew B. Davies[1]

Departments: [1]Department of Zoology and Entomology, University of Pretoria, Pretoria, 0002, South Africa

[2]School of Biological Sciences, The University of Queensland, QLD 4072, Australia

[3]Frankfurt Zoological Society – Bale Mountains Conservation Project, PO Box 265, Bale-Robe, Ethiopia

Degree: Master of Science (Zoology)

Abstract

Although protected areas have been used as principal conservation tools, most of them are suffering from human-induced threats. Consequently, a good understanding of such human-driven threats on biodiversity and identifying early warning systems for habitat change in protected areas is necessary for effective conservation of natural resources. To examine the impact of human disturbance on avifaunal assemblages and to assess the potential application of birds as bioindicators of forest health monitoring in the Afromontane forest of the Bale Mountains of Ethiopia, I recorded birds and habitat variables in three protected and three unprotected forests using a point transect method in 2009 and 2012. The two land use types differ in disturbance levels (higher in the unprotected areas), vegetation structure and bird assemblages. Species richness of entire bird guild, open woodland and open land habitat guilds, granivore and insectivore feeding guilds, and shrub layer and ground layer foraging substrate guilds of birds were significantly higher in the unprotected areas than the protected areas. Abundances of guilds of birds mostly followed a similar trend with species richness. However, densities of overall and forest-specialist bird guilds were higher in the protected area and *vice versa* for the other guilds. In general, the protected area assemblages were dominated by forest-specialist species, while those of the unprotected areas were dominated by openland and shrubland species. The implication is that disturbance had caused encroachment of non-native species (openland, open woodland and shrub land species) while negatively affecting native species (forest species, particularly tree canopy foragers). These assemblage differences are linked to changes in vegetation structure caused by disturbance. Thus, further forest degradation in the protected area should be avoided in order to maintain native/forest-specialist species. Given the differences in bird assemblages between the two land use types, there is a high likelihood that bioindicator species (i.e. indicator species - those 'characteristic' of a particular habitat - and detector species - those occurring in the different habitats considered but with moderate indication value) can be identified, therefore providing a useful tool to monitor ecosystem health of the forests. Four and nine species were identified as appropriate indicator species (i.e. species with indicator values > 60% and fulfilling biological and niche history criteria used in selection) in the protected and unprotected areas, respectively. In addition, nine species were identified as detectors of habitat change in the protected areas. These bioindicators provide a useful tool for managers of Afromontane forest in the Bale Mountains, as well as similar habitats elsewhere, for long term monitoring of ecosystem health of the forests.

Keywords: Afromontane forest, avifaunal assemblages, avifaunal guilds, Bale Mountains, bioindicators, birds, detector species, disturbance, indicator species, protected area

Acknowledgements

I am heartedly thankful to my supervisors Professor Andrew McKechnie, Professor Berndt J. van Rensburg, Dr Anouska Kinahan and Dr Andrew Davies for your tireless contribution you made towards the completion of this dissertation. I am lucky enough to be supervised by such enthusiastic scientists [you]; your guidance, knowledge transfer and encouragements to learn new ideas is always much appreciated. Your comments and suggestions on the proposal and dissertation were not only crucial to its completion, but also were educational courses to me, for this I am deeply grateful to all of you. My sincerer thanks go to Professor Andrew for your support in fund-raising, my placement as an MSc student at the University of Pretoria and making travel arrangements. I would like to thank Professor Berndt specifically for the training provided on ecological data analyses techniques. Dr Anouska, deserves especial thanks not only for facilitating my field works in Ethiopia and trips to South Africa, but also you were always there for me the time when I was frustrated and felt hopeless; you were the source to get my hope and morale revived and without your encouragement and support in this regard this work would not be completed. Dr Andrew Davies, I am greatly indebted to you, as a supervisor and friend, for your being always with me in the office and patiently listening to supporting me on matters ranging from silly to challenging ones.

Several people were involved as assistant during the field work. I thank you Usman Abdella, Jemal Kassim, Jemal Abduku, Jemal Mohammed and Aman Guto for your tireless and

consistence assistance during the work even under harshly environmental conditions. Bale Mountains National Park drivers, Mesfin Tafesse and Guluma Tilehun, thank you for the help provided in driving me and my assistants to and from field sites. Thadaigh Baggally was supporting me in facilitating financial sources and logistics for my field work in Ethiopia, for this thank you very much.

A number of friends and relatives had provided their support in one way or the other way. My friends in the Ethiopian Wildlife Conservation Authority, Kahsay Gebretensae and Rezenom Almaw, had shared the suffering I passed through to get study permit due to bureaucratic system in that office. Without your heartedly support and encouragement, this work would have not been completed. Thank you so much for your being there for me during those bad days. Dr Anteneh Shimelis had provided his comments on draft version of chapter one of the dissertation –many thanks go to you. Many people were encouraging me in the course of study; *inter alia*, Daniel Tilaye and his families, Chris Gordon, Anne-Merie Stuart, Professor Claudio Sillero-Zubiri, Dereje Tadesse, Girma Mengesha, Mohamed Welu, Gebeyehu Likassa, Edris Ebu, Zegeye Kibret, Tesfaye Tolla, Abebayehu Kahsaye, Abde Amano, Husen Hiko, Mustefa Dulle, Muzeyen Turkie, Mohamednur Kebeto, Hordefa Beyecha, Issa Hasan, Safia Hadji, Ayuba Ahmed, Aschalew Beyene, Geremew Mebratu, Abdurazak Adem, Fatuma Alo and Ahmed Alo. I would like to thank you for this. I would like to thank Ingrid Vis for your help during registration and accessing to student bursaries at the University of Pretoria.

My families and relatives were always with me during the work towards this dissertation. Asratu Asefa, Tadesse Asefa, Endalew Endegena, Bilisa Endegena and Tesfaye Asefa, as well your families, thank you all. My mother, Badhane Ayana (who passed away in 2008), and my father, Asefa Mitiku, deserve especially thanks for letting me see the light of science.

Chadzamira families in Pretoria are duly acknowledged for providing me a room in your flat and making my stay there a home-away-from-home. Your encouragements during my study and familial life I spent with you are all unforgettable.

This study was undertaken with financial support from DST/NRF Centre of Excellence at the Percy FitzPatrick Institute /University of Pretoria and Frankfurt Zoological Society-Bale Mountains Conservation Project. The University of Pretoria is also acknowledged for offering me the opportunity to study as an MSc studentship position. Thanks to the Ethiopian Wildlife Conservation Authority/Bale Mountains National Park for the study permission given and to the Embassy of the Republic of South Africa for issuing me study permit visa.

Disclaimer

Styles and formats, as well citations and bibliographies are standardized across all chapters of this dissertation. However, some overlap in content may occur throughout the dissertation to ensure ease of understanding and flow. For ease of reading, tables and figures have been embedded at appropriate positions in each chapter.

Table of Contents

Content **Page**

Chapter 1: General introduction

Background and justification

Protected areas (e.g., national parks, sanctuaries and reserves) remain the principal tool for the *in situ* conservation of biodiversity and unique ecosystems (Wynne 1998; Locke and Dearden 2005; Gaston et al. 2008; Jackson et al. 2009; Game et al. 2013). However, as a result of increased anthropogenic threats, poor management systems and limited finances, many protected areas are in danger of not achieving the specific conservation goals for which they were originally set aside or proclaimed (van Schaik et al. 1997; Struhsaker et al. 2005; Bleher et al. 2006). This is particularly true in developing countries, where ever-increasing human populations threaten protected areas through activities incompatible with conservation (Bruner et al. 2001; Struhsaker 2001). For instance, a survey conducted in 201 parks across 16 tropical countries revealed that more than 70% of these parks are affected by poaching, clearing, encroachment and/or logging (van Schaik et al. 1997; see also Bruner et al. 2001 for similar results in other tropical countries). An important step to mitigate these impacts in protected areas in order to enhance the conservation of biological diversity is for conservation managers and policy makers to have detailed information on the range of impacts within these areas (Bruner et al. 2001; Struhsaker 2001; Bleher et al. 2007).

Studies have shown that human disturbances in forest ecosystems generally have negative consequences for biodiversity (Şekercioğlu 2002; Chown 2010; Mengesha et al. 2011). However, fine scale studies have found conflicting results; in some areas, protected natural habitats tend to contain higher species richness and/or abundance of particular biological taxa than the surrounding unprotected or disturbed habitats (e.g. Recher 1969; Trzcinski et al. 1999; Heikkinen et al. 2004; Kessler et al. 2005), while other studies have documented the opposite response (e.g. see Kumar & Ram 2005, for plants; Tabeni and Ojeda 2005, for rodents; Gove et al. 2013, for birds). Similar results have been reported when a given animal community is grouped by functional guild. For instance, Canaday (1997) studied the impact of disturbance on birds in Amazonian rainforest in Ecuador and found a reduced number of insectivores in areas of greater human impact. In contrast, Gove et al. (2008) found higher bird species richness in disturbed sites than relatively intact montane forest in Ethiopia. This implies that the impact of habitat disturbance on biodiversity can be either positive or negative, depending on the type and severity of the disturbance, study area and the taxa considered. Although some anthropogenic disturbances to forests result in the reduction of vegetation cover and consequently habitat for some biological taxa (e.g. forest specialist mammals and birds) with negative biodiversity consequences, it can also be a means of creating habitat heterogeneity for generalists and other groups of animals such as some invasive species (Fahrig 2003; van Rensburg et al. 2009). Therefore, a detailed understanding of the impacts on biodiversity of different human disturbances, and the underlying processes, is needed for protected area managers to practice effective conservation management activities. This involves documenting the responses of well known animal groups such as birds (Kati & Şekercioğlu 2006) and/or functional guilds (e.g. insectivore and frugivore birds) and relating their responses to key habitat features.

In addition, the identification of indicator species that characterize a particular habitat and its status is becoming popular among biodiversity conservationists and environmentalists (Niemi & McDonald 2004). Indicator taxa possess an undeniable appeal for conservationists, land managers, and governments as they provide a cost- and time-efficient means to assess the impacts of environmental disturbances on biodiversity and ecosystems (Carignan & Villard 2002; Niemi & McDonald 2004). Given the limited budgets and expertise allocated to many protected areas in developing countries (Struhsaker et al. 2005) identifying and monitoring such biological indicator species is indispensable for protected area managers to make scientifically informed management decisions (Kati & Şekercioğlu 2006; OARDB 2007). Birds are suggested to be ideal taxa for applying the indicator concept, on account of their sensitivity to environmental changes, relatively well understood biology, and ease of identification and sampling (Bock & Webb 1984; Kati & Şekercioğlu 2006)

In this study, I examine the impact of anthropogenic habitat disturbance on birds by comparing avifaunal assemblages in protected and unprotected dry Afromontane forests in the northern Bale Mountains of southeast Ethiopia (Figure 1). The greater Bale Mountains region is generally recognized as a biodiversity hotspot by Conservation International (Williams et al. 2004). At the heart of these mountains is the Bale Mountains National Park (BMNP), proclaimed in 1971 to conserve the ecological and hydrological systems, and the rare, threatened and endemic floral and faunal species of the mountains (Hillman 1986, 1988). The BMNP is recognized as a centre of diversity and endemism, where many of the plant and animal species are also locally endemic. About 78 mammal, 300 bird and 17 amphibian species, of which 17, 6 and 11, respectively, are endemic, are known to occur in the BMNP (Asefa 2006/07, 2011; Largen & Spawls 2011). It is identified as an important bird area (IBA), harbouring seven globally threatened species and 88% of Ethiopia's highland biome species (EWNHS 1996). The national park also protects part of the largest pieces of Afro-alpine habitat on the African continent (Yalden 1983) as well as the hydrological systems of the Bale massif on which the livelihoods of over 12 million people depend (Hillman 1986, 1988; Williams et al. 2004;

OARDB 2007). However, this rich biodiversity area with vital ecosystem services is deteriorating due to ever-increasing human and livestock encroachment (Stephens et al. 2001; Williams et al. 2004; OARDB 2007; Abera & Kinahan 2011; Teshome et al. 2011).

The Bale Mountains area was virtually uninhabited prior to the 1960's, before the establishment of the BMNP (Brown 1969; Hillman 1986; Miehe & Miehe 1994; Teshome et al. 2011). Since its designation, BMNP has, like many of Africa's protected areas (Struhsaker et al. 2005), come under increasing pressure from an ever-growing human population in the surrounding area (Stephens et al. 2001; NH 2004; Williams et al. 2004; OARDB 2007). Consequently, settlement, subsistence cultivation, livestock grazing, selective logging and deliberate forest fires have been increasingly threatening biodiversity and ecosystems within and around the BMNP (Table 1; Hillman 1986; Miehe & Miehe 1994; Stephens et al. 2001; OARDB 2007; Abera & Kinahan 2011; Assefa 2011; Teshome et al. 2011). A lack of human and financial resources, political interest and technical knowledge has contributed to the decline in management effectiveness and the consequent degradation of BMNP (OARDB 2007).

The northern slopes of the BMNP are covered by patches of dry Afromontane forest vegetation, in which *Juniperus procera*, *Hagenia abyssinica* and *Hypericum revoluteum* are the characteristic tree species. These forest patches, six in number, vary in size and have been under different land use since the establishment of the BMNP in 1971 (Table 1). Three of these patches (Adellay, Boditti and Dinsho Hill [BMNP HQs] forests) are entirely included in the BMNP, while the other three (Angesso, Shaya and Darkina forests) are only partially incorporated within the park (Figure 1; Hillman 1986; OARDB 2007). Moreover, due to a lack of manpower and financial input, only the three patches that are completely embedded within the park boundary are under effective protection; the remaining three are unprotected with free access for human settlement, agriculture, firewood collection and livestock grazing (Hillman 1986; Stephens et al. 2001; OARDB 2007). It is therefore expected that the forests in these different land use categories differ in the abundance, diversity and assemblages of bird species they support. This is because changes in land use result in adverse impacts on some ecological components of the forests while favouring others (Laiolo et al. 2004; Martin & Possingham 2005; Gove et al. 2008). However, avifaunal studies in the Bale Mountains have been no more than checklist-based until recently (Asefa 2006/07; Shimelis et al. 2011; Shimelis et al. *in prep.*). This remains the current status of avian data, despite the 10-year General Management Plan of the BMNP stating the need for the collection of baseline avian data, especially within the northern forests given their important conservation status. The identification of indicator species that could act as an early warning of habitat change has also been identified as a priority research topic (OARDB 2007). Indicator species should be monitored over time thereby informing managers of the national park about the ecosystem health of the forests (OARDB 2007). This approach is based on the assumption that birds are key indicators of ecosystem health and productivity and can be easily monitored with minimum inputs (Carignan & Villard 2002; Niemi & McDonald 2004; Kati & Şekercioğlu 2006).

The overall objective of this study is to better understand how mechanisms related to anthropogenic disturbances that account for changes in vegetation structure may translate into altered patterns in the distribution of birds associated with Afromontane forest in the northern Bale Mountains, southeast Ethiopia. More specifically, this study will examine differences in bird abundance, diversity and assemblages between the two land use types, which will enable indicator species to be identified so that ecosystem health and biodiversity can be monitored over time. This study therefore will not only provide valuable information on globally important habitats and species, but also locally relevant information by providing a baseline for the implementation of long term monitoring of ecosystem health and biodiversity in the BMNP.

Study area

The Bale Mountains region is located 400 km southeast of Addis Ababa, the capital of Ethiopia. It belongs to the Bale-Arsi massif, which forms the western section of the south-eastern Ethiopian highlands. At the centre of these mountains is the BMNP (717163- 789904 N, 552126-604402 E; Fig. 1). The current extent of the national park is 2200 km^2, encompassing a landscape that ranges in altitude from 1500 to 4377 m a.s.l. (Hillman 1986). The area experiences two rainy seasons, known locally as the heavy and small rains. The heavy rains last from July to October, with a peak in August, while the small rains last from March to June, peaking in April. Records show that this area experiences temperature extremities during the dry season. The lowest recorded temperature on the highest plateau in Bale (Sanetti) is -15º C, while it is -6º C on the northern slope; the recorded maximum is 26º C (OBARD 2007). Five broad vegetation zones occur in the BMNP and the surrounding areas, namely the northern grasslands (a flat area at an altitude of 3000 m a.s.l.), the northern dry Afromontane forest (2900-3400 m a.s.l.), ericaceous forest (3400-3800 m a.s.l.), the Afro-alpine moorland and grassland (3800-4377 m a.s.l.), and the southern Harenna forest (1500-3200 m a.s.l.; Hillman 1986; Miehe & Miehe 1996; NH 2004; OARDB 2007).

In addition to its large size and separation from the rest of the Ethiopian highlands (Yalden 1983), the prevalent rain (eight months per year), varying topography, and diverse habitats in the Bale Mountains have provided the isolation necessary for the evolution of distinct animal and plant species (Williams et al. 2004; Asefa 2011; Largen & Spawls 2011). Due to the complexity of the region, the area is known for its high levels of species richness and endemism spanning a range of taxonomic groups contributing to the overall biological importance of the region from both an ecological and evolutionary perspective. For example, 17 (22%) of the seventy-eight mammal species and six (2%) of the 278 bird species recorded in the region are endemic to the country (Asefa 2006/07, 2011). Five of the mammal species are currently presumed to be confined to the Bale Mountains area, while another five species have been reported from only few localities outside the BMNP (Yalden 1988; Asefa 2011). In addition, of 17 amphibian species belonging to 7 genera recorded, 14 of the species (with 5 species locally endemic) and 5 of the genera (with 3 of them locally endemic genera) are Ethiopian endemics (Williams et al. 2004; Largen & Spawls 2011). The Bale Mountains region contains 1600 species of flowering plants of which 160 and 34, are Ethiopian endemic and locally endemic respectively (NH 2004; Williams et al. 2004). The region is important not only for the conservation of such high floral and faunal species diversity (many of which are significant from a global conservation standpoint), but it also plays a crucial role in maintaining the hydrological systems of the area. Over 40 streams originate from the mountains, and are vital to the survival of over 10 million people dwelling in the south-eastern lowlands of Ethiopia and northern Somalia (OARDB 2007).

Despite its immense importance as a centre of endemism and evolutionary processes, this region is currently under conservation threat (Hillman, 1986; Stephens et al. 2001; OARDB 2007; Abera & Kinahan 2011; Assefa 2011; Teshome et al. 2011). Agricultural expansion, livestock grazing, deliberate forest fire, logging for timber production and settlement expansion are increasing at an unprecedented rate. These human activities in the region pose severe threats to the survival of wildlife and the ecosystem of the region (OARDB 2007).

The present study was conducted in the northern dry Afromontane forest of the Bale Mountains. There are a number of reasons why this forest type was selected for this study. First, it represents a critical habitat for the globally endangered endemic mountain nyala (*Tragelaphus buxtoni*, containing about two-thirds of the entire global population of this species) and other ungulates as well as several endemic birds (Hillman 1986, 1988; Refera & Bekele 2002; OARDB 2007; Asefa 2008). Second, being located at the northern limit of the BMNP, it is experiencing unprecedented human pressure (see Hillman, 1986; OARDB 2007; Abera & Kinahan 2011 for more information). Finally, it is a relatively less studied habitat type in the Bale Mountains, and no quantitative information exists on the avifauna of the forest (Asefa 2006/07; Shimelis et al. 2011). The forest’s vegetation is dominated by two indigenous tree species, *Hagenia abyssinica* and *Juniperus procera*, with *Hypericum revoltum* and *Rappanea simensis* also abundant at higher elevations (Hillman 1986; NH 2004). The northern dry Afromontane forest occurs in six isolated patches varying in altitude between 3100-3450 m a.s.l. Due to the spatial location of these forest patches relative to protected area infrastructure and manpower, such as the amount of patrolling activities, some of these patches are under more pressure to human activities than others (Table 1; see also Hillman 1986, 1988; OARDB 2007; Teshome et al. 2011 for more details). As a result, these forest patches can be characterized into two different land use classes, namely protected (Adellay, Boditti and Dinsho Hill/BMNP HQs) and unprotected (Angesso, Shaya and Darkina).

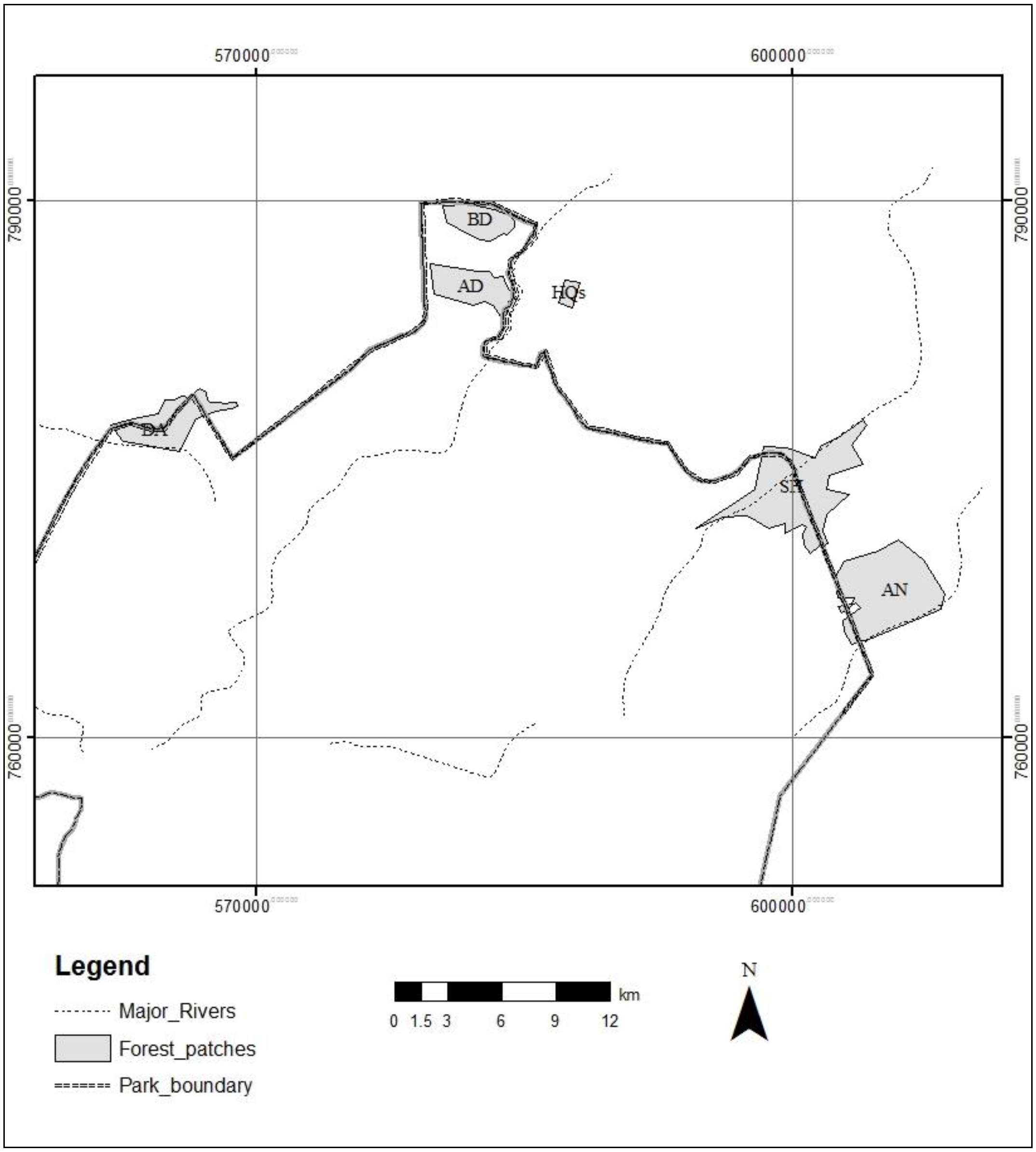


Figure 1. Map of the Bale Mountains National Park (only partly shown) and the six forest patches studied. Abbreviations of the patches: AN = Angesso; SH = Shaya; HQs = Dinsho Hill (BMNP HQs); AD Adellay; BD = Boditti; and DA = Darkina.

Table 1. Description of the study forest patches and their threat levels.

Patch Name	Protection status	Area (ha)	Distance to the nearest patch (km)	Patrolling effort (man-hour/day)	Settlement (mean no. houses/ha)	Logging (mean no. stumps/ha)	Grazing level	Agriculture (mean % cover/ha)
Adellay	Protected	784.9	1.6	40	0	60.0 ± 15.0	None-low	0
Boditti	Protected	555.1	1.6	40	0	27.5 ± 2.5	Low-moderate	0
Dinsho Hill (BMNP HQs)	Protected	118.9	2.5	40	0	17.5 ± 12.5	None-low	0
Angesso	Unprotected	2237.9	1.4	0.4[a]	1.2 ± 0.1	122.5 ± 12.5	Moderate-Heay	42.1 ± 12.1
Darikna	Unprotected	1016.6	15.3	0.4[a]	4.5 ± 0.2	182.5 ± 32.5	Moderate-Heay	67.1 ± 10.8
Shaya	Unprotected	2485.1	1.4	0.4[a]	2.4 ± 0.3	190 ± 32.5	Moderate-Heay	3.4 ± 1.4

[a] – These unprotected forest patches are sporadically monitored (on the average four times each year by fie people for eight hours each time, which is equal to about 0.4 man-hour per day)

Structure of the dissertation

Following this introduction, Chapter 2 examines the impacts of human disturbance on forest bird communities by comparing bird species richness, population abundance and assemblage composition in protected and unprotected Afromontane forest patches, and explores habitat variables that might explain the observed patterns. Chapter 3 deals with the selection of bird species that could act as bioindicators of ecosystem health of the forests for long term monitoring. Finally, Chapter 4 presents the general conclusions of the overall dissertation, focusing on management and conservation implications and provides recommendations for future research.

References

Abera, K., and A.A. Kinahan. 2011. Factors Affecting Fire Extent and Frequency in the Bale Mountains National Park. Walia-Special Edition on the Bale Mountains 146-157.

Asefa, A. 2006/07. Birds of Bale Mountains National Park, Southeast Ethiopia. Walia **25**: 22-33.

Asefa, A. 2008. Mountain Nyala and Ethiopian Wolf Mortalities in the northern side of Bale Mountains National Park, Ethiopia. Ethiopian Journal of Biological Sciences **7**: 179-184.

Asefa, A. 2011. Mammals of the Bale Mountains National Park, Ethiopia: a compiled and annotated checklist. Walia-Special Edition on the Bale Mountains 3-14.

Bleher, B., D. Uster, and T. Bergsdorf. 2006. Assessment of threat status and management effectiveness in Kakamega Forest, Kenya. Biodiversity and Conservation **15**:1159–1177.

Bock, C.E., and B. Webb. 1984. Birds as grazing indicator species in the southeastern Arizona. Journal of Wildlife Management **48**: 1045-1049

Bruner, A.G., R.E. Gullison, R.E. Rice, and G.A.B. da Fonseca. 2001. Effectiveness of Parks in Protecting Tropical Biodiversity. Science **291**:125-127

Brown, L H. 1969. Observations on the status, habitat and behaviour of the mountain nyala *Tragelaphus buxtoni* in Ethiopia. Mammalia **33**: 545-597.

Canaday, C. 1997. Loss of insectivorous birds along a gradient of human impact in Amazonia. Biological Conservation **77**: 63-77.

Carignan, V. & M. Villard. 2002. Selecting indicator species to monitor ecological integrity: a review. Environmental Monitoring and Assessment **78**: 45–61.

Chown, S.L. 2010. Temporal biodiversity change in transformed landscapes: a southern African perspective. Philosophical Transactions of the Royal Society, Biological sciences **365**: 3729-3742.

EWNHS. 1996. *Important Bird Areas of Ethiopia: a first inventory*. Ethiopian Wildlife and Natural History Society, Addis Ababa, Ethiopia.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual Review in Ecology Evolution and Systematics **34**: 487–515.

Game, E.T., Kareiva, P. and Possingham, H.P. 2013. Six Common Mistakes in Conservation Priority Setting. Conservation biology *Conservation Biology* DOI: 10.1111/cobi.12051*s*.

Gaston, K. J., S.F. Jackson, L. Cantu-Salazar, and G. Cruz-Pinon. 2008. The Ecological Performance of Protected Areas. Annual Review Ecology Evolution and Systematics **39**: 93–113.

Gove, A.D., K. Hylander, S. Nemomisa, & A. Shimelis. 2008. Ethiopian coffee cultivation—Implications for bird conservation and environmental certification. Conservation Letters **1**: 208–216.

Gove, A.D., K. Hylander, S. Nemomsa, A. Shimelis, and W. Enkossa. 2013. Structurally complex farms support high avian functional diversity in tropical montane Ethiopia. Journal of Tropical Ecology **29**: 87-97.

Heikkine, R., M. Luoto, R. Virkkala, & K. Rainio, 2004. Effects of habitat cover, landscape structure and spatial variables on the abundance of birds in agricultural-forest mosaic. Journal of Applied Ecology **41**: 824-835.

Hillman, J.C. 1986. Bale Mountains National Park Management Plan. Ethiopian wildlife conservation organization, Addis Ababa, Ethiopia

Hillman, J.C. 1988. The Bale Mountains National Park area, southeast Ethiopia, and its Management. Mountain Research and Development **8**: 253-258.

Jackson, S.F., K. Walker, and K.J. Gaston. 2009. Relationship between distributions of threatened plants and protected areas in Britain. Biological Conservation **142**: 1515–1522.

Kati, V.I. and C.H. Şekercioğlu. 2006. Diversity, ecological structure, and conservation of the landbird community of Dadia reserve, Greece. Diversity and Distributions **12**: 620–629

Kessler, M., P.J.A. Kebler, S.R. Grandstein, K. Bach, M. Schmill, and R. Pitopang. 2005. Tree diversity in primary forest and different land use systems in central Sulawes, Indonesia. Biodiversity and Conservation **14**: 547-560.

Kumar, A. & J. Ram. 2005. Anthropogenic disturbances and plant biodiversity in forests of Uttaranchal, Central Himalaya. Biodiversity and Conservation **14**: 309-331.

Laiolo, P., F. Dondero, E. Ciliento, and A. Rolando. 2004. Consequences of pastoral abandonment for the structure and diversity of the alpine avifauna. Journal Applied Ecology **41**: 294- 304

Largen, M., and S. Spawls. 2011. Amphibians and Reptiles Recorded from the Bale Mountains. Walia-Special Edition on the Bale Mountains 89–91.

Locke, H., and P. Dearden. 2005. Rethinking protected area categories and the new paradigm. Environmental Conservation **32**: 1–10.

Martin, T.G., and H.P. Possingham. 2005. Predicting the impact of livestock grazing on birds using foraging height data. Journal of Applied Ecology **42**:400-408.

Miehe, G. & S. Miehe. 1994. Ericaceous forest and heath land in Bale Mountains of South Ethiopia, Ecology and Man‘s Impact. Traut Warnke Verlag, Hamburg, Germany.

Mengesha, G., Y. Mamo, and A. Bekele. 2011. A comparison of terrestrial bird community structure in the undisturbed and disturbed areas of the Abijata Shalla lakes national park, Ethiopia. International Journal of Biodiversity and Conservation **3**: 389-404.

NH (The National Herbarium). 2004. Biodiversity assessment of the Bale Mountains National Parkand surrounding areas. The National Herbarium of Addis Ababa University, Addis Ababa, Ethiopia.

Niemi, G.J., and M.E. McDonald. 2004 Application of ecological indicators. Annual Review in Ecology Evolution and Systematics **35**: 89-111

OARDB 2007. Bale Mountains National Park General Management Plan. Oromia Agriculture and Rural Development Bureau, Oromia National Regional State, Ethiopia.

Recher, H.F. 1969. Bird species diversity in Australia and North America. The American Naturalist **103**: 75-80.

Refera, B., and A. Bekele. 2002. Diurnal activity pattern of Mountain Nyala (*Tragelaphus buxtoni)* in Bale Mountains National Park, Ethiopia. Ethiopian Journal of Biological Sciences **1**: 63-72.

Şekercioğlu, C.H. 2002. Effects of forestry practices on vegetation structure and bird community of Kibale National Park, Uganda. Biological Conservation **107**: 229–240

Shimelis, A., A. Bekele, A. Asefa , S. Wiliams, A. Gove, and S. Thirgood. 2011. Structuring of the Birds of the Bale Mountains National Park. Walia-Special Edition on the Bale Mountains 15-27.

Stephens, P.A., C.A. D'sa, C. Sillero-Zubri, and N.L. Williams. 2001. Impact of livestock and settlement on the large mammalian wildlife of Bale Mountains National Park, Southern Ethiopia. Biological Conservation **100**: 307-322.

Struhsaker, T.T., P.J. Struhsaker, and K.S. Siex. 2005. Conserving Africa's rain forests: problems in protected areas and possible solutions. Biological Conservation **123**: 45-54.

Tabeni, S., and R.A. Ojeda. 2005. Ecology of the Monte Desert small mammals in disturbed and undisturbed habitats. Journal of Arid Environments **63**: 244-255

Tadese, D., S. Wiliams, and B. Irwin. 2011. People in National Parks – Joint Natural Resource Management in Bale Mountains National Park – Why it makes sense to work with local people. Walia-Special Edition on the Bale Mountains 257-268.

Teshome, E., D.Randal, and A.A. Kinahan. 2011. The Changing Face of the Bale Mountains National Park over 32 years: A Study of Land Cover Change. Walia-Special Edition on the Bale Mountains 118-130.

Tews, J., U. Brose, V. Grimm, K. Tielbörger, M. C. Wichmann, M. Schwager1 and F. Jeltsch 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. Journal of Biogeography **31**:79–92.

Trscinski, M.K., L. Fahrig, & G. Merriam. 1999. Independent effects of forest cover and fragmentation on the distribution of forest breeding birds. Ecological applications **9**: 586-593.

van Rensburg, B.J., S.L. Chown, A.S. van Jaarsveld, & M.A. McGeoch. 2000. Spatial variation and biogeography of sand forest avian assemblages in South Africa. Journal of Biogeography **27:**1385–1401

van Rensburg, B.J., D.S. Peacock, & M.P. Robertson. 2009. Biotic homogenization and alien bird species along an urban gradient in South Africa. Landscape and Urban Planning **92**: 233–241

van Schaik C.P., J. Terborgh and D. Dugelby. 1997. The silent crisis: the state of the rainforest nature reserves. *In*: Kramer R., van Schaik C.P. and Johnson J. (eds), The Last Stand: Protected Areas and the Defense of Tropical Biodiversity. Oxford University Press, New York, pp. 64–89.

Williams, S., J.L. Vivero Pol, S. Spawls, A. Shimelis, and E. Kelbessa. 2004. Ethiopian Highlands. In: *Hotspots revisited* (Mittermeier, R.A., Gill, P.R., Hoffmann, M., Pilgrim, J., Brooks, T., Mittermeier, C.G., Lamoreux, J. and Da Fonseca, G.A.B. (eds.)). CEMEX Publisher, Behington.

Wynne, G., 1998. Conservation policy and politics. In: Sutherland, W.J. (ed.), Conservation, Science and Action. Blackwell Science, Oxford, pp. 256-285.

Yalden, D.W. 1983. The extent of high ground in Ethiopia compared to the rest of Africa. SINET: Ethiopian Journal of Sciences **6**: 256-285.

Chapter 2

Humans and Afromontane birds in the Bale Mountains of Ethiopia: how severe are the impacts?

Abstract*:* *Although protected areas have been used as principal conservation tools, most of them are suffering from human-induced threats. For these reasons, a good understanding of the overall anthropogenic threats associated with a lack of protection to biodiversity in protected areas is necessary for effective conservation of natural resources. To examine the impact of human disturbance on avifaunal assemblages in the Afromontane forest of the Bale Mountains of Ethiopia, I recorded birds and habitat variables in three protected forests and three unprotected forests using point transects in 2009 and 2012. The two land use types differ in disturbance levels (being higher in the unprotected areas), vegetation structure and bird assemblages. Species richness of (1) entire bird assemblage, (2) woodland and open land habitat guilds, (3) granivore and insectivore feeding guilds, and (4) shrub layer and ground layer foraging substrate guilds of birds were significantly higher in the unprotected areas compared to the protected areas. Abundances of most of these guilds followed a similar trend as species richness. However, the result that current disturbance levels lead to an increase in overall bird species richness compared to less disturbed and protected forests should be interpreted with caution. This is because densities for all species examined, forest-specialists and those species that forage in the tree canopy were higher in the protected areas compared to unprotected. Similarly and in addition, the relative contribution of each guild to the overall richness and abundance of the assemblages showed that the forest specialists and those species foraging in the canopy layer were more pronounced in the protected areas; and conversely, woodland-, shrubland- and openland-specialists together with granivores were dominant in the unprotected areas. This suggests that forest-specialist species are negatively affected by habitat disturbance and non-forest-specialist species are favoured. Thus, further forest degradation in the protected areas should be avoided in order to conserve native forest-specialist bird species.*

Keywords: Afromontane forest, avifaunal assemblages, Bale Mountains, disturbance, protected area, avifaunal guilds.

2.1 Introduction

The degradation of tropical forests and destruction of habitats due to anthropogenic activities are major causes of decline in global biodiversity (Brooks et al. 2006). In attempts to mitigate these effects, protected areas (e.g. national parks, sanctuaries and reserves) have been used as principal conservation tools (Wynne 1998; Locke & Dearden 2005; Gaston et al. 2008; Jackson et al. 2009). However, many protected areas are in danger of being unable to effectively conserve the biodiversity priorities and ecological values for which they were originally proclaimed. This is true particularly in developing countries that are often characterised by rapid human population growth rates leading to agricultural intensification, an increase in livestock overgrazing, logging and settlement encroachments. These factors are typically among the main drivers of habitat degradation identified globally (Millennium Ecosystem Assessment 2005; Chown 2010). Other factors include poor management systems, limited finances and political instability (van Schaik et al. 1997; Bruner et al. 2001; Struhsaker et al. 2005; Bleher et al. 2006). Understanding these impacts, especially within protected areas, and the way in which plants and animals respond to them, is an important step for efficient conservation decision making (Bruner et al. 2001; Bleher et al. 2007; Game et al. 2013).

Although the consequences of human disturbances in forest ecosystems are generally presumed to have negative consequences on biodiversity (Şekercioğlu 2002; Chown 2010), studies have shown conflicting results. In some areas protected natural habitats show a tendency to contain higher species richness and/or abundance of particular biological taxa than the surrounding unprotected or disturbed habitats (Recher 1969; Trzcinski et al. 1999; Heikkinen et al. 2004; Mengesha et al. 2011, for birds; Kessler et al. 2005, for plants), whereas at other localities the opposite pattern has been documented (e.g. Kumar & Ram 2005, for plants; Tabeni and Ojeda 2005, for rodents; Gove et al. 2013 for birds). Similar results have been reported when a given animal community is grouped into functional guilds. For instance, Canaday (1997) studied the impact of disturbance on birds in Amazonian rainforest in Ecuador and found reduced numbers of insectivores in areas of greater human impact. Şekercioğlu et al. (2002) also documented the disappearance of insectivorous birds from tropical forest fragments near Las Cruces, southern Costa Rica. In contrast, Gove et al. (2008) found higher bird species richness in disturbed sites than relatively intact sites of montane forest in Ethiopia. This implies that the impact of habitat disturbance on biodiversity can be either positive or negative, depending on the type and severity of the disturbance and the biota or biotic group considered (Chown 2010). While some human-induced disturbances to forests result in a reduction of vegetation cover and subsequently habitat for some biological taxa (e.g. forest specialist mammals and birds) with negative consequences, it can also be a means of creating habitat heterogeneity for generalists and other groups of animals, including invasive species (Fahrig 2003; van Rensburg et al. 2009).

Birds are particularly susceptible to habitat destruction and alteration due to changes in the availability and abundance of food as well as nesting and safe-resting sites, with profound impacts on avifaunal assemblages (Martin 1984; Vickery et al. 2001; Martin & Possingham 2005; Mengesha et al. 2011; Sinclair et al. 2002). Reduction of vegetation cover due to overgrazing or deforestation and changes in soil properties from trampling or agricultural intensification can have severe impacts on vegetation structure and composition and subsequently bird assemblages (Jensen 1985; Kerley & Whitford 2000; Woldu & Mohammed

Saleem 2000; Vickery et al. 2001; Chown 2010; Sinclair et al. 2002). However, as discussed above, such changes in assemblages may be positive or negative. For example, frugivorous birds have been found to disappear from heavily transformed forest landscapes, but at the same time granivorous species have increased in species richness and abundance in forests converted to agricultural land (Gove et al. 2008; Gove et al. 2013). Thus, any activity which results in habitat modification has the potential to significantly impact bird assemblages.

The Bale Mountains massif is considered a biodiversity hotspot by Conservation International (Williams et al. 2004). The Bale Mountains National Park, the core conservation area of this massif is currently on the tentative list for World Heritage site listing, and is an Important Bird Area of Ethiopia (EWNHS 1996). Hosting nearly 300 bird species, the Bale Mountains harbour six of the 18 Ethiopian endemics and an additional 14 near endemic (Ethiopia/Eritrea) bird species (Asefa 2007/2007; Shimelis et al. 2011). Thus, from both a biodiversity and economic (tourism industry) perspective, Bale's avifaunal assemblages are important to the area and its local communities. However, due to increasing population pressure, land is being heavily altered and degraded mainly through agricultural and settlement expansion, overgrazing and deforestation (FZS/BMNP, *unpublished data*), all of which could have a profound effect on the avifaunal assemblages found in the area. As such, understanding any changes to bird species composition and identifying the drivers of these changes is imperative for effective management and mitigation of potential negative consequences on these important assemblages.

In this study, I examine the impact of human disturbance on avifaunal composition by comparing assemblages in protected and unprotected Afromontane forests in the northern Bale Mountains of southeast Ethiopia. Specifically, I examine patterns in bird diversity (species richness, population density, and the composition of avian assemblages) in these different land use types (protected and unprotected), and the potential underlying mechanisms related to changes in vegetation structure as a result of human disturbance. Human-induced threats are found to be widespread in all the forest patches (Akele 2007; OARDB 2007), but are heavier and more heterogeneous in the unprotected than in the protected forests. Thus, it is predicted that the protected forests contain higher species richness and abundance (density) of overall birds, forest habitat guilds and canopy layer foraging guilds than the unprotected forests. In contrast, as a result of the opening up of habitat and crop cultivation, it is predicted that the unprotected forests host higher species richness and abundance (density) of openland, woodland and shrubby habitat species guilds, granivore feeding guilds and shrub layer and ground layer foraging guilds than the protected forests.

2.2 Materials and Methods

2.2.1 Study area

The Bale Mountains region is located in southeast Ethiopia and belongs to the Bale-Arsi massif, which forms part of the Ethiopian highlands. At the centre of these mountains is the Bale Mountains National Park (BMNP; located at 717163 - 789904 N, 552126 - 604402 E; see also Figure 1 in Chapter 1). The park covers 2200 km^2 and encompasses the largest expanse of Afro-alpine habitat on the African continent and Ethiopia’s second largest tropical montane forest. It ranges in altitude from 1500 to 4377 m a.s.l. (Hillman 1986). The area experiences two rainy seasons, the heavy rainy season is from July to October and the small rains last from March to June with a dry season from November to February, mean annual rainfall is 1219 mm. The area experiences temperature extremes during the dry season and has a mean annual minimum and maximum temperature of 2.36° C and 15.5° C, respectively (Hillman 1986). Five broad vegetation zones occur in the BMNP and surrounding areas, namely the northern grasslands (a flat area at an altitude of 3000 m a.s.l.), the northern dry Afromontane forest (2900 - 3400 m a.s.l.), ericaceous forest (3400 - 3800 m a.s.l.), the Afro-alpine moorland and grassland (3800 - 4377 m a.s.l.), and the southern Harenna forest (1500 - 3200 m asl) (Hillman 1986; Miehe & Miehe 1996; NH 2004; OARDB 2007).

Due to its isolation from the rest of the Ethiopian highlands (Yalden 1983) and the complexity of its habitat, distinct animal and plant species have evolved in the Bale Mountains and there are consequently very high levels of species richness and endemism (Williams et al. 2004; Asefa 2011; Largen & Spawls 2011). Despite its immense importance as a centre of endemism and evolutionary processes, the region is currently under conservation threat due to settlement, crop production and livestock grazing driven by a growing human population (Hillman 1986; Stephens et al. 2001; OARDB 2007; Abera & Kinahan 2011; Assefa 2011; Teshome et al. 2011).

The northern part of the BMNP and its surrounding areas comprises patches of dry afromontane forest, characterised by *Juniperus procera*, *Hagenia abysinica* and *Hypericum revoluteum* tree species. This study was conducted in six of these forest patches of varying size (120 - 2485 ha) and level of anthropogenic threat (Table 1 of Chapter 1), but similar altitude (2900 – 3400 m a.s.l.), dominant vegetation and topography. Three patches (Adelay, Boditti and Dinsho Hill) lie inside the “protected” national park and three partly outside (Angesso, Shaya and Darkina) in “unprotected” adjacent areas (Figure 1 of Chapter 1).

2.2.2. Bird surveys

In each of the six dry Afromontane forest patches, five line transects of 1 to 1.5 km were randomly placed parallel to each other along on an altitudinal gradient of 2900 – 3400 m a.s.l., and spaced a minimum of 300 m from each other to reduce pseudo-replication.

Along each line transect, four fixed bird survey points were selected and point counts carried out. Point transect counting is considered more suitable for sampling cryptic, shy and skulking species in forest habitats where detection probability is reduced by dense vegetation cover, and to relate bird occurrences with habitat features (Gibbons et al. 1996; Bibby et al. 1998; Gregory et al. 2004; Buckland et al. 2001). Points along each transect were selected systematically to avoid effects of habitat edges and double counting between adjacent points. The first fixed points were selected by ensuring a minimum of 50 m distance from the edge of the forest boundary, and the distance between two adjacent points was limited within a range of 200 m – 350 m. By using transect start and end GPS points it was ensured that the same transects were resampled over time.

Bird surveys were carried out in the dry (November to March) and wet (July to October) seasons of 2009 and 2012. Each transect in all patches was surveyed twice on a given day (early morning and late afternoon), thus, each transect was visited eight times in the course of the study. Several authors (e.g. Gibbons et al. 1996; Bibby et al. 1998) have recommended a sampling period of seven to ten minutes, depending on the type of habitat and birds surveyed. Thus, a sampling period of eight minutes was used in the present study with an additional two minutes allocated for birds to settle before commencing counting at each point. Within the eight minutes, birds seen and/or heard within a radius of 50 m were recorded along with their number and estimated sighting distance (in bands of 5 m intervals). Birds flushed away from the census point while approaching the station and those that flew away while counting were recorded from the point they were first seen (van Rensburg et al. 2000; Gregory et al. 2004). Birds that were seen flying over the census area and not necessarily making use of the habitat (e.g., swifts, swallows, scavengers and some raptors) were not recorded. However, some predatory birds that hunt from tree canopies (e.g., African Goshawk and Little Sparrow-hawk) and seen making use of the habitat at the time of observation were recorded**.** Counts were conducted early in the morning, between 07h30 and10h30, and late in the afternoon, between 14h30 and17h30 when the majority of birds are active.

2.2.3. Measurements of habitat variables and land degradation

In addition to avian data, vegetation structure and disturbance variables were recorded for each forest patch at the same points where point counts were conducted. Vegetation structure variables recorded were: tree abundance (woody species with height > 3m), percentage canopy cover (recorded to the nearest 5%), canopy height (to the nearest 1 m), shrub height (to the nearest 5 cm), percentage cover of shrubs (woody plants with height < 3 m) and height and percentage basal cover of forbs and grass. For trees, sampling was within quadrats of 20 m × 20 m set up at each of the bird counting points. Abundance of each tree species was counted within the quadrates and canopy height measured using a Sunto Clinometer (Newton 2007). Canopy cover was visually estimated; at each quadrat four readings from the four cardinal directions were taken at the centre and the average of the four readings for a given quadrat was calculated and used as the percentage canopy cover of that quadrat (Newton 2007). At the corners of each of the 20 m × 20 m quadrats, four 5 m × 5 m sub-quadrats were established to quantify shrub, forb and grass height and percentage cover. Heights of these plant growth forms (shrub, forbs and grass) were measured using a 3 m high labelled measuring stick and percentage cover was visually estimated. For each of the plant growth forms, height was determined by randomly taking four different measurements within each of the four sub-quadrats (totalling 16 measurements at each sampling point). At the centre of the four 5 m × 5 m sub-quadrats established at each bird counting plot, another 2 m × 2 m sub-quadrat was established to visually estimate the proportion of bare ground. The average of all readings from the sub-quadrats were calculated and used as the height of each vegetation growth form for that quadrat. The visually estimated percentage cover of shrubs, forbs and grasses as well as bare ground within the four sub-quadrats at each main quadrat was also averaged and used as the percentage cover of each plant growth form.

In each of the 20 m x 20 m quadrats the number of tree stumps (from firewood harvesting or logging) was counted and the grazing levels estimated using an ordinal scale as follows: 0 = no grazing; 1 = light; 2 = moderate; and 3 = heavy grazing, based on the indicators of different grazing pressures proposed by Newton (2007). Lastly, at each of these sampling points, within a radius of 50 m, the proportion of visible land under cultivation was estimated and the number of houses present counted.

2.3 Data Analysis

Since six patches (three of each type) only were available for sampling, transects were used as sampling units to overcome bias due to smaller size of using patches as replicates. However, the transects in each forest patch might not be independent of each other, potentially resulting in type I error. Therefore, the results should be interpreted keeping these limitations in mind.

2.3.1 Forest disturbance and vegetation structure

Since cultivation and settlement were only encountered in the unprotected sites, comparisons between land use types were investigated based only on the density of tree logging and the level of grazing. The mean values for grazing and tree stump abundance were obtained for each transect and compared between protected and unprotected forest patches. Mann Whitney *U*-tests were used for comparisons since data were non-normally distributed. SPSS version 20 software (IBM Corporation 2011) was used for these analyses.

Data on tree abundance collected at each quadrat along each transect in each season over the two years were summed and used as tree abundance for that transect; while values for heights and percentage cover of trees, shrubs, grass and forbs and percentage coverage of bare ground were averaged and used as representative values for each transect in each season. These values were square-root transformed and normalized prior to analysis in order to standardize measurement scales. Analysis of Similarity (ANOSIM) was used to examine differences between the two land use types in terms of the habitat variables based on the multivariate data set (Clarke & Gorley 2006). Euclidean distance, an appropriate measure for environmental data, was used to measure the similarity between treatments (Clarke & Gorley 2006). In addition, one-way ANOA was conducted in SPSS software based on averaged values of the transformed data from the two seasons to see which of the variables specifically characterize each of the land use types.

2.3.2 Bird species richness

Sample-based rarefaction curves, calculating species density values (the number of species per unit area), were compiled for the two land use types across seasons to establish sampling representivity (Gotelli and Colwell 2001). The observed rarefaction curves were calculated using a moment-based interpolation method (Mao Tau) with EstimateS v. 8.2 (http://viceroy.eeb.uconn.edu/estimates; Colwell 2009). Sampling is considered to be adequate if the observed rarefaction curve approaches an asymptote (Longino et al. 2002) or if it converges closely with an appropriate richness estimator. The Jacknife2 richness estimator was used because, unlike most other estimators, it does not require species to be independent or sampling points to be compositionally similar and data can have either a parametric or non-parametric distribution (Chao 2004; Magurran 2004), and the index provides conservative but accurate richness estimates (Magurran 2004). The estimated richness values for each land use were calculated without resampling as this approach produces more accurate richness estimates compared to a sampling approach (Colwell 2009).

To compare the species richness values between land use types, rarefaction curves must be scaled by number of individuals (calculating species richness) and not by number of samples (calculating species density). The sample-based data were therefore scaled by the number of individuals from which individual-based rarefaction curves were computed in EstimateS using the Coleman method (Coleman 1981). Thereafter, for each land use comparison examined for the dry season, wet season, and seasons combined, the curves were rarefied to the lowest number of individuals recorded in a given land use type to ensure valid comparisons of species richness between land use types (Gotelli & Colwell 2001).

Prior to analysis, bird assemblages were classified into three different guild types based on requirements for preferred habitat, feeding substrate, and diet. For the preferred habitat classification, bird assemblages were assigned to one of four major habitat groups following the classification of Urban & Brown (1971), Sinclair & Ryan (2003) and Redman et al. (2009): (1)

open habitat species (fields and grazing areas), (2) shrubland species (shrubby areas, forest edge habitats and crop fields), (3) woodland species (wooded savanna and farmlands with scattered trees and sub-urban parks), and (4) forest species (dense, closed forests/woodlands). For the classification related to feeding substrates, three categories were used, namely (1) tree canopy (species using lower, mid and/or upper tree canopy), (2) shrub layer (species using forest under story and/or shrubbery vegetation in shrub land areas), and (3) ground layer (species foraging on the forest floor or open ground). Finally, following the methods of Kissling (2011) and Gove et al. (2013) and based on dietary requirements, bird assemblages were assigned to one of six broad feeding guilds: (1) frugivore, (2) granivore, (3) insectivore, (4) nectarivore, (5) carnivore and (6) omnivore.

Species richness (estimated based on Jacknife2) and abundances (number of individuals recorded) for all species (seasons separately) and guilds of species (seasons combined due to small sample size for most of the guilds) were compared between the protected and unprotected areas using Analysis of Variance in SPSS version 20 software (IBM Corporation 2011).

2.3.3 Bird population density

Density estimations and modelling of the detection function (i.e., decreasing probability of detection with increasing distance from the observer) were performed using DISTANCE 4.1 Release 2 software (Thomas et al. 2003), with post-stratification performed across the different subsets of guilds established above for each land use type across seasons. Unlike traditional methods of density estimation, distance sampling analysis takes into account the differences in detectability of animals due to seasonal and/or habitat heterogeneity (Bibby et al. 1998; Buckland et al. 2001; Thomas et al. 2003). Prior to fitting an appropriate detection function, a 10 percent right truncation, i.e., discarding observations beyond 45 m, was applied as recommended by Buckland et al. (2001). The remaining distances were split in to four intervals (0 - 10 m, 11 – 20 m, 21 - 35 m and 36 - 45 m). These interval bands and selection of different intervals were selected based on their best fit compared to other interval options examined. Observations of conspecific bird flocks comprising more than a single individual were treated as clusters of individuals. Several detection function models with different adjustment terms were used as suggested by Thomas et al. (2003). The best model was selected based on Akaike Information Criterion (AIC) values. In these analyses, uniform key with cosine adjustment was selected in all cases except in the protected area during the wet season where half-normal with polynomial adjustment was selected. Densities were obtained for the entire bird community (number of birds.ha^{-1}) and for guilds of species defined above for each land use type across seasons and pooled data.

2.3.4 Assemblage composition

A multivariate approach was implemented using PRIMER software (Clarke & Gorley 2006) to assess variation in bird species composition between land use types using abundance data obtained for every species along each transect. A Bray-Curtis similarity index was used to calculate similarities in composition among assemblages, data were fourth-root transformed beforehand to down-weight common species relative to those that are rare (Clarke & Warwick 1994). Global R values were used to determine the degree of similarity among treatments. This is a non-parametric permutation procedure applied to rank similarity matrices underlying sample ordinations (Clarke 1993). The closer the value of R value is to 1, the more dissimilar species assemblages are. In addition, similarity percentage analysis (SIMPER) was conducted in

PRIMER software to calculate the percentage contribution each species made to the dissimilarities between the two assemblages across seasons (Clarke & Gorley 2006).

2.3.5 Bird–habitat relationships

The summed and averaged values for each habitat variable from the replicated surveys were calculated for each transect and square-root transformed and normalized. Bird abundance patterns were then matched to habitat variables separately for each of the two land use types across seasons, with the bird species and habitat data matrices as the input according to PRIMER's BEST procedure (Clarke & Gorley 2006). The BEST procedure is used to find the 'best' match between the multivariate community analysis among sample patterns of an assemblage and that from the environmental variables associated with those samples; the extent to which these two patterns match reflects the degree to which the chosen habitat variables explain the assemblage pattern (Clarke & Gorley 2006). Variable selection was made using a Bio-Env algorithm, which searches all possible combinations from the primary datasheet. Spearman rank correlation (P_s) was used to measure the strength of the relationship between the bird and habitat resemblance matrices (Clarke & Gorley 2006).

2.4. Results

2.4.1 Disturbance and vegetation structure

Of the four major disturbance parameters assessed, only two, namely tree logging and livestock grazing, were encountered in both land use types examined. Crop production and settlement were only recorded in the unprotected areas. Of the 15 transects sampled in the unprotected areas, houses were encountered along seven of them (47%) and crop cultivation along nine

(60%) with mean values of 2.7 houses and 40% cover.ha^{-1}, respectively. The number of stumps occurring was significantly higher in the unprotected areas than the protected areas (number of tree stumps.ha^{-1}: protected area median = 14.06, unprotected area median = 84.38; $U = 3.500$, $P < 0.0001$). Similarly, grazing was found to be significantly higher in the unprotected forests compared to the protected forest patches, with median values of 2.5 (i.e. moderate-heavy grazing) and 0.5 (i.e. none to light grazing), respectively (wet season: Mann-Whitney $U = 5.00$; dry season: 7.50, $P < 0.0001$ for both seasons). However, differences within sites for seasonal grazing levels were only detected in the unprotected areas (unprotected area Mann-Whitney $U =$ 57.5, $P = 0$ .021; protected area Mann-Whitney $U = 98.5$, $P = 0.567$), implying that some forest patches are more heavily disturbed than others in unprotected areas, whereas all protected ones experience similar levels of disturbance.

Significant seasonal differences in habitat variables were observed both within and between land use types, except during the wet season between protected and unprotected areas when differences were less pronounced (Table 1, Fig. 1). The ANOSIM results showed that, considering pair-wise tests of all possible comparisons, the strongest differences in habitat variables were recorded between protected area dry and wet seasons (Table 1).

Table 1. Analysis of similarity (ANOSIM) for habitat variables within and between the protected areas (PA) and unprotected areas (UPA) during wet and dry seasons

Groups	PA (wet) vs UPA (wet)	PA (wet) vs. PA (dry)	PA (dry) vs UPA (dry)	UPA (dry) vs UPA (wet)
R Statistic	0.228	0.746	0.407	0.439
Significance Level %	0.1	0.1	0.1	0.1

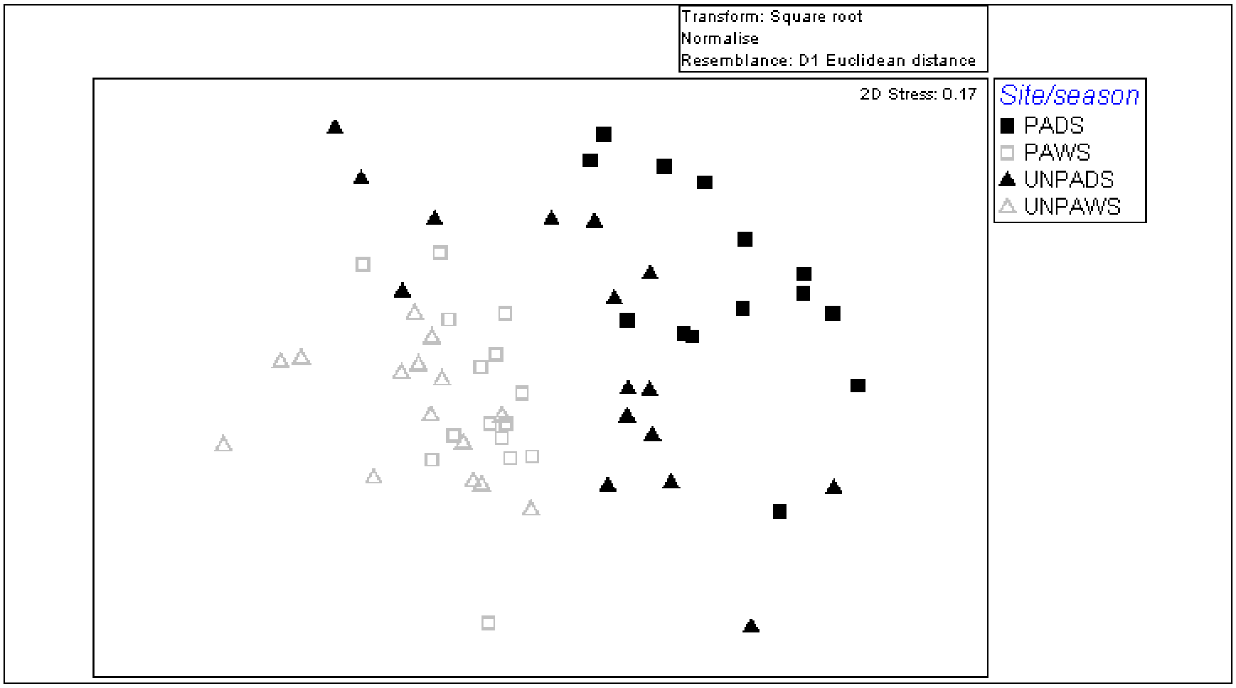


Figure 1. Non-metric multi-dimensional scaling ordination of habitat variables in the protected and unprotected afromontane forest of the BMNP across wet and dry seasons (PADS = protected area dry season, PAWS = protected area wet season, UNPADS = unprotected area dry season, UNPAWS = unprotected area wet season).

Levene's test of homogeneity of variances indicated that three variables (tree cover, shrub cover and grass height) did not meet this assumption (in all cases Levene's Statistic = 4.362-7.993, d.f. = 1, 28, $P < 0.05$), and the results of non-parametric test did not change the decision of the test. Thus, the results from ANOVA were reported here. In general, the results of ANOVA analysis showed that abundance and percentage cover of trees, and height and percentage cover of grass were significantly higher in the protected area than in the unprotected area (in all cases, F = 4.78-35.15; d.f. = 1, 28; $P < 0.05$), and *vice-versa* in the case of percentage cover of bare ground (F = 15.77, d.f. = 1, 28, $P < 0.05$; Table 2).

Table2. Results of One-way ANOVA analysis comparing means (square-root transformed) of each habitat variable between the protected area (PA) and unprotected area (UPA).

Variable	Area	Mean ± S.E.	ANOVA					
			Sources of variation	SS	df	MS	F	Sig.
Tree abundance	PA	5.1 ± 0.3	Between Groups	34.62	1	34.62	31.28	0.000
	UPA	3.0 ± 0.2	Within Groups	30.99	28	1.11		
			Total	65.60	29			
Tree height (m)	PA	4.4 ± 0.2	Between Groups	0.10	1	0.10	0.24	0.625
	UPA	4.3 ± 0.2	Within Groups	11.27	28	0.40		
			Total	11.37	29			
% tree cover	PA	6.2 ± 0.2	Between Groups	38.21	1	38.21	35.15	0.000
	UPA	4.0 ± 0.3	Within Groups	30.44	28	1.09		
			Total	68.65	29			
Shrub height (m)	PA	1.4 ± 0.2	Between Groups	0.01	1	0.01	0.01	0.905
	UPA	1.3 ± 0.1	Within Groups	10.16	28	0.36		
			Total	10.17	29			
% shrub cover	PA	3.3 ± 0.3	Between Groups	8.10	1	8.10	2.25	0.145
	UPA	4.4 ± 0.6	Within Groups	100.69	28	3.60		
			Total	108.78	29			
Grass height (cm)	PA	4.3 ± 0.2	Between Groups	14.80	1	14.80	24.83	0.000
	UPA	2.9 ± 0.1	Within Groups	16.68	28	0.60		
			Total	31.48	29			

% grass cover	PA	6.7 ± 0.3	Between Groups	5.47	1	5.47	4.78	0.037
	UPA	5.9 ± 0.2	Within Groups	32.00	28	1.14		
			Total	37.47	29			
Forbs height (cm)	PA	3.9 ± 0.2	Between Groups	0.04	1	0.04	0.03	0.866
	UPA	3.8 ± 0.3	Within Groups	34.14	28	1.22		
			Total	34.18	29			
% forbs cover	PA	6.0 ± 0.3	Between Groups	3.41	1	3.41	1.96	0.172
	UPA	5.3 ± 0.4	Within Groups	48.64	28	1.74		
			Total	52.05	29			
% bare ground	PA	3.5 ± 0.3	Between Groups	23.24	1	23.24	15.77	0.000
	UPA	5.3 ± 0.3	Within Groups	41.26	28	1.47		
			Total	64.50	29			

2.4.2 Bird species richness

Although sample-based rarefaction curves started to flatten off for both land use types (Fig. 2 a-c), these curves did not reach an asymptote nor did they converge closely with the observed Jacknife2 richness estimator. The observed species richness values should therefore be treated with caution and as such site comparisons between land use types were made based on individual-based rarefaction curves (Coleman rarefaction) and Jacknife2 richness estimates. Overall, 6 477 observations (2 951 in the protected area and 3 526 in the unprotected area) were recorded. Of the total 66 species recorded in both land use types during both seasons, 60 species (52 during dry season and 53 during wet season) and 46 species (41 during dry season and 39 during wet season) were recorded in the unprotected area and the protected area, respectively (Appendix I). Estimated bird species richness (Jacknife2) was significantly higher in the unprotected areas than the protected areas during the wet season, and when seasonal data were pooled for each land use type ($F_{1,28} = 9.989$, $P = 0.004$ for wet season; $F_{1,28} = 4.995$, $P = 0.034$ for pooled data; Table 3). Although species richness was still higher in the unprotected areas than the protected areas during the dry season, this difference was not significant ($F_{1,28} = 2.562$, $P = 0.121$). Individual-based rarefaction curves (Coleman rarefaction curves) also showed similar trends for each season, as well when seasons were combined (Fig. 3a-c). Considering the 13 subsets of the three sets of guilds, a significantly higher number of bird species were recorded in the unprotected areas than the protected areas for woodland and open land habitat guilds (woodland: $F_{1,28} = 69.340$, $P < 0.0001$; openland: $F_{1,28} = 30.791$, $P < 0.0001$), granivore ($F_{1,28} = 16.344$, $P < 0.0001$) and insectivore ($F_{1,28} = 5.304$, $P = 0.029$) feeding guilds, and shrub ($F_{1,28} = 7.979$, $P = 0.009$) and ground layer ($F_{1,28} = 22.905$, $P < 0.0001$) foraging substrate guilds (Figs. 4-6).

Table 3. Overall estimated avian species richness (based on Jacknife2 estimates) and their mean abundance (based on raw abundance data) in the protected and unprotected areas during the dry season, wet season and pooled season (15 samples in each land use type; F and *P* values are tests for difference in richness and abundance between the two land use types in each season; significance level, alpha = 0.05).

Season	Habitat	Richness (mean ± S.E.)	Abundance (mean ± S.E.)	s	S
		($F_{1,28}$ = 2.562, *P* = 0.121)	($F_{1,28}$ = 1.043, *P* = 0.316)		
Dry season	Protected area	42.41±3.39	3013.87 ± 435.01	53	41
	Unprotected area	50.84±4.03	3719.47 ± 536.86	52	52
		($F_{1,28}$ = 9.989, P = 0.004)	($F_{1,28}$ = 1.110, *P* = 0.301)		
Wet season	Protected area	37.66±3.37	1613.87 ± 232.94	50	39
	Unprotected area	57.20±5.18	2005.33 ± 289.44	70	52
		($F_{1,28}$ = 4.995 P = 0.034)	($F_{1,28}$ = 0.967, *P* = 0.334)		
Pooled season	Protected area	46.98±4.49	1400.00 ± 202.07	63	46
	Unprotected area	61.53±4.75	1714.13 ± 247.41	70	59

s = rarefied species richness (individual-based; Coleman method, S = estimated species richness (Jacknife2).

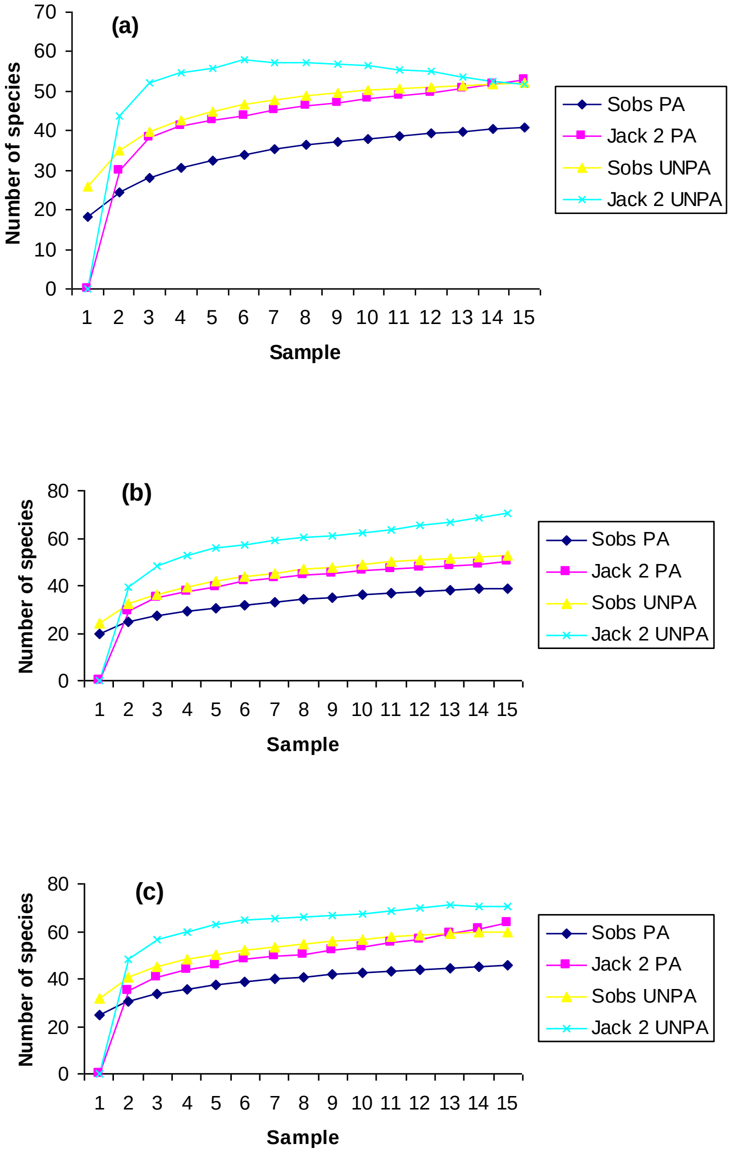


Figure 2**.** Sample-based (Mao-Tau) avian species rarefaction and richness estimator curves based on the observed number of species (Sobs) and Jacknife 2 richness estimator, respectively, for protected areas (PA) and unprotected areas (UNPA) during (a) the dry season, (b) the wet season, and (c) when seasons are pooled.

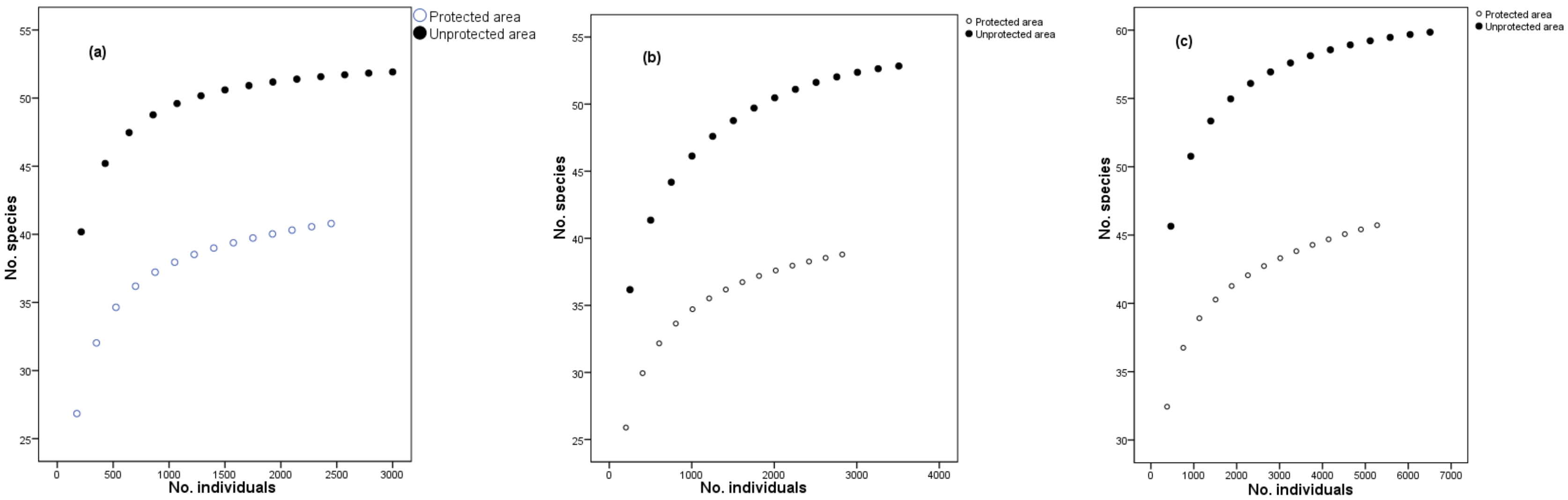


Figure 3. Individual-based (Coleman rarefaction curves) avian species rarefaction curves based on the observed number of individuals for protected area (○) and unprotected area (●) for (a) the dry season, (b) the wet season, and (c) when season data are pooled.

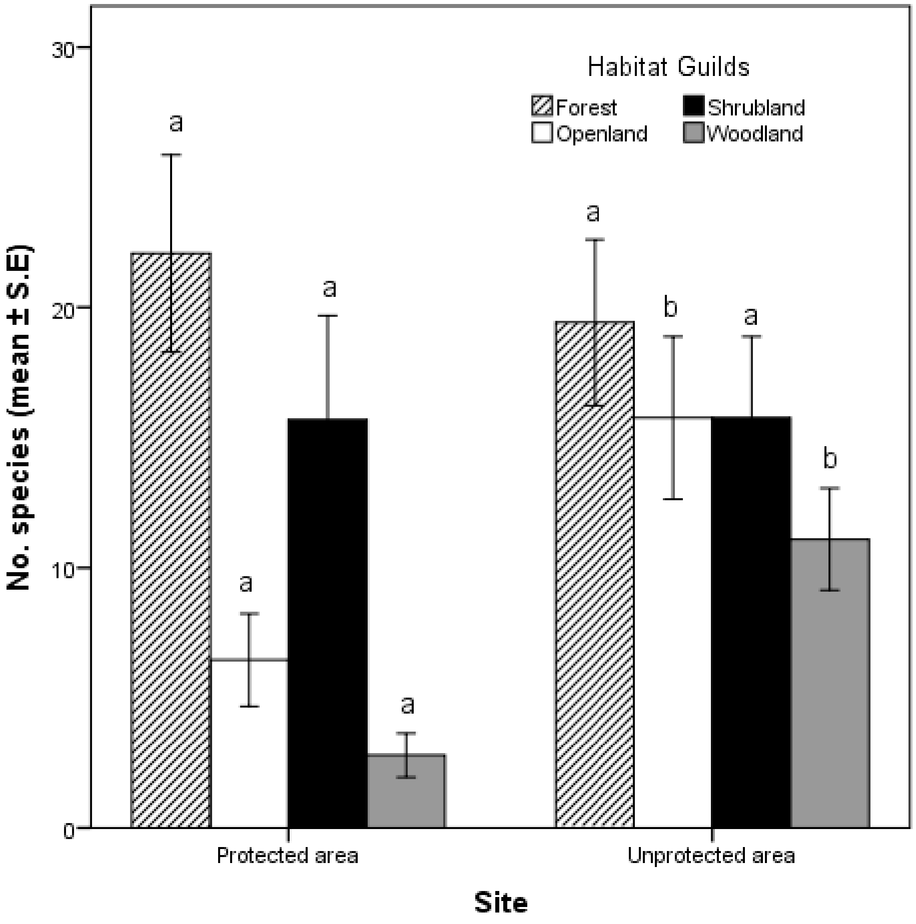


Figure 4. Mean (± 1 SE; based on Jackknife2 estimates) species richness of avian habitat guilds in the protected and unprotected areas for seasons combined. Similar letters in each series denote guilds which are the same statistically between sites; differences are considered significant at the α-level of 0.05.

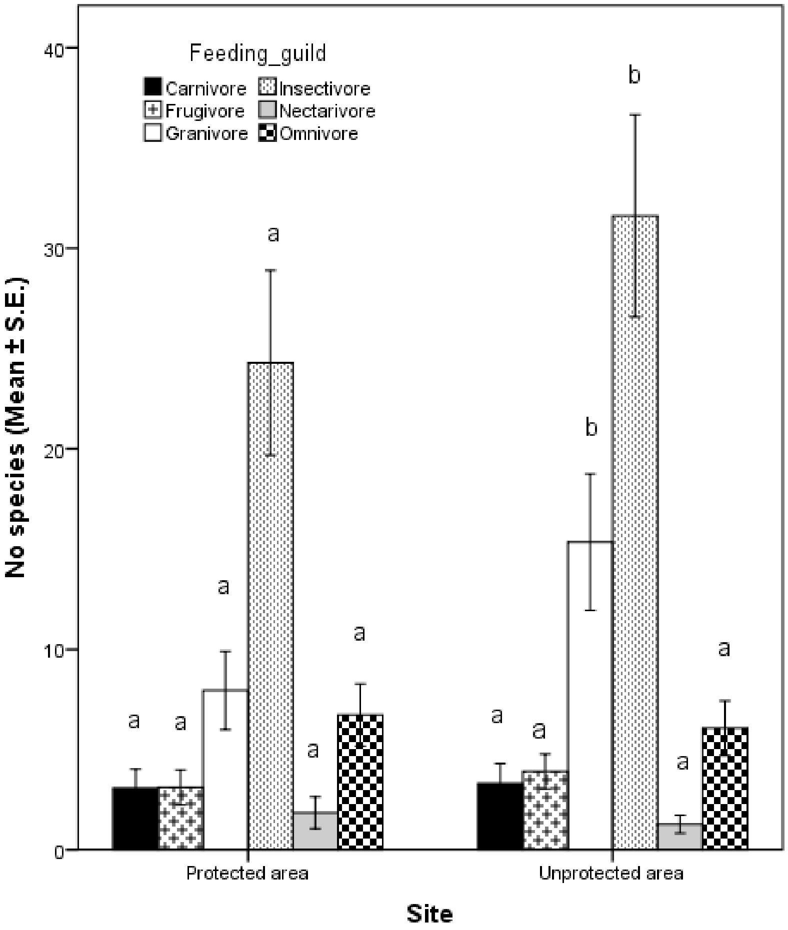


Figure 5. Mean (± 1 SE; based on Jacknife2 estimates) species richness of avian feeding guilds in the protected and unprotected areas for seasons combined. Similar letters in each series denote guilds which are the same statistically between sites; differences are considered significant at the α-level of 0.05.

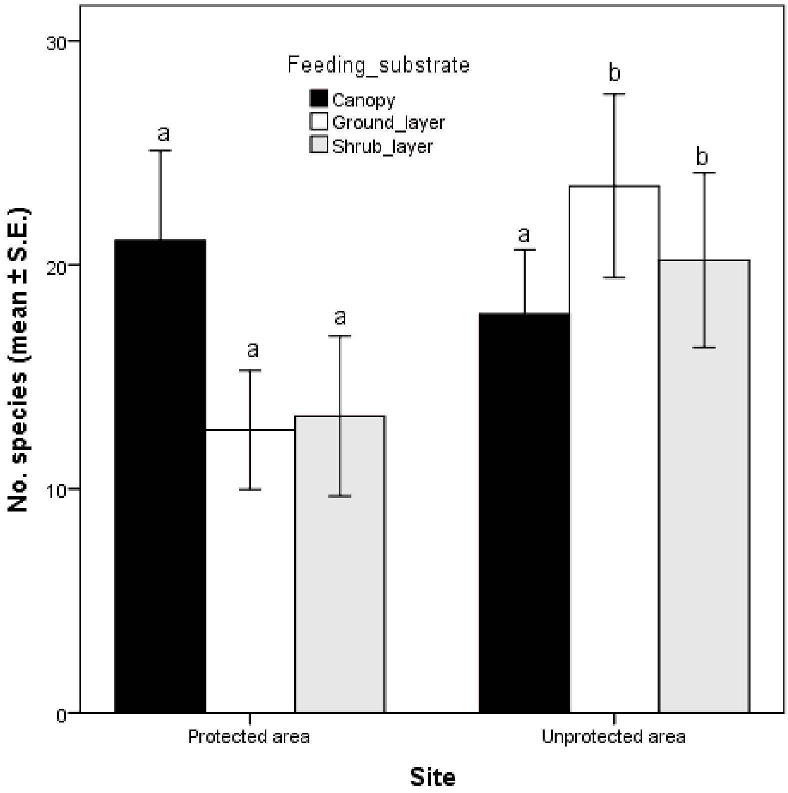


Figure 6. Mean (± 1 SE; based on Jacknife2 estimates) species richness of avian feeding substrate guilds in the protected and unprotected areas for seasons combined. Similar letters in each series denote guilds which are the same statistically between sites; differences are considered significant at the α-level of 0.05.

2.4.3 Bird abundance

Overall bird abundance was not significantly different between the two land use types either when seasons were treated separately or when combined (ANOVA: dry season, $F_{1,28} = 1.043$, $P = 0.316$; wet season, $F_{1,28} = 1.110$, $P = 0.301$; pooled season $F_{1,28} = 0.967$, $P = 0.334$; Table 3). However, abundances of avian guilds followed a similar trend as species richness; most of the guilds with higher species richness in the unprotected areas also had higher abundances than in the protected areas (Figs. 7-9). Of the habitat-associated guilds, woodland ($F_{1,28} = 5.246$, $P = 0.030$) and openland ($F_{1,28} = 22.615$, $P < 0.0001$) guilds had significantly higher abundances in the unprotected areas than the protected areas. A significant difference in abundance was recorded for granivore birds, with higher numbers in the unprotected areas than the protected areas ($F_{1,28} = 16.206$, $P < 0.0001$). Of the 13 subsets of guilds the only guild with significantly higher abundance in the protected areas was the canopy substrate foraging guild ($F_{1,28} = 5.310$, $P = 0.029$; Figs. 7-9).

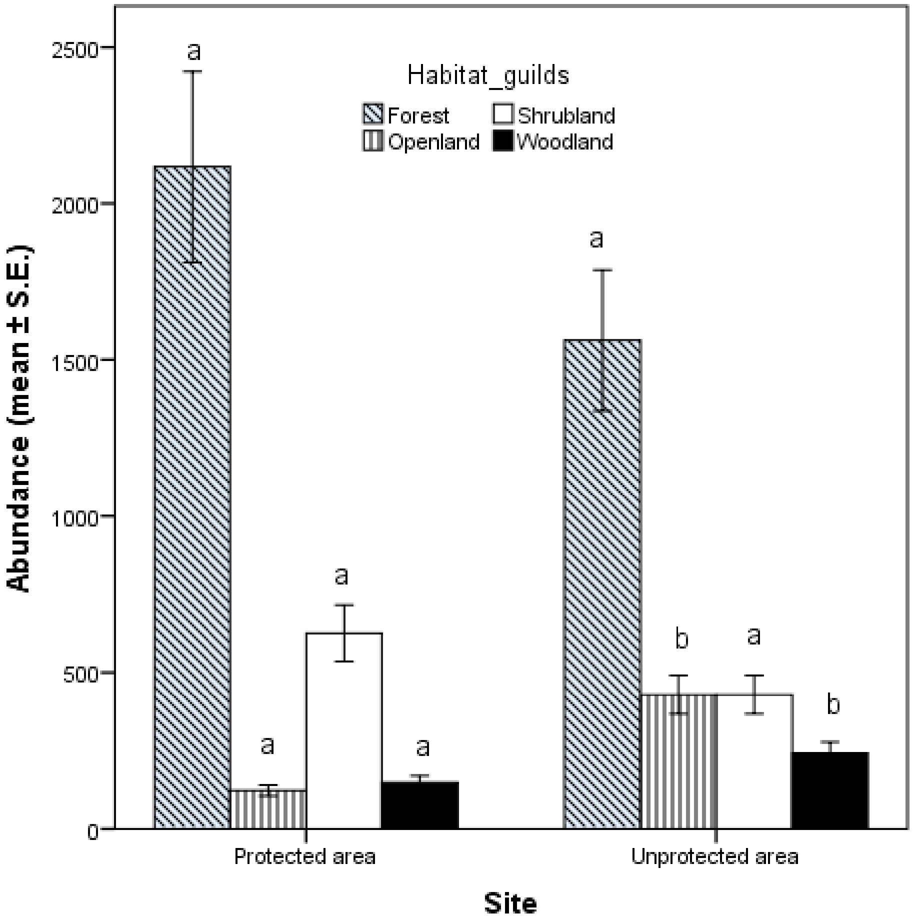


Figure 7. Mean (± 1; SE) abundance of avian habitat guilds in the protected and unprotected areas for seasons combined. Similar letters in each series denote guilds which are the same statistically between sites; differences are considered significant at the α-level of 0.05.

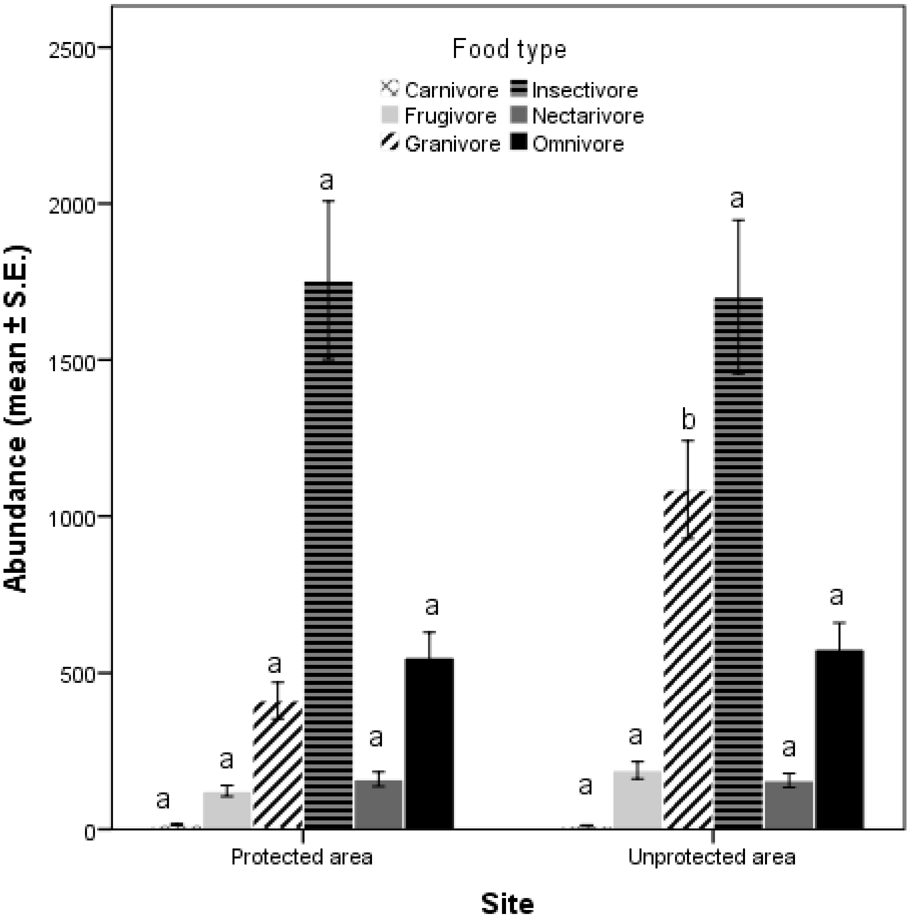


Figure 8. Mean (± 1 SE) abundance of avian feeding guilds in the protected and unprotected areas for seasons combined. Similar letters in each series denote guilds which are the same statistically between sites; differences are considered significant at the α-level of 0.05.

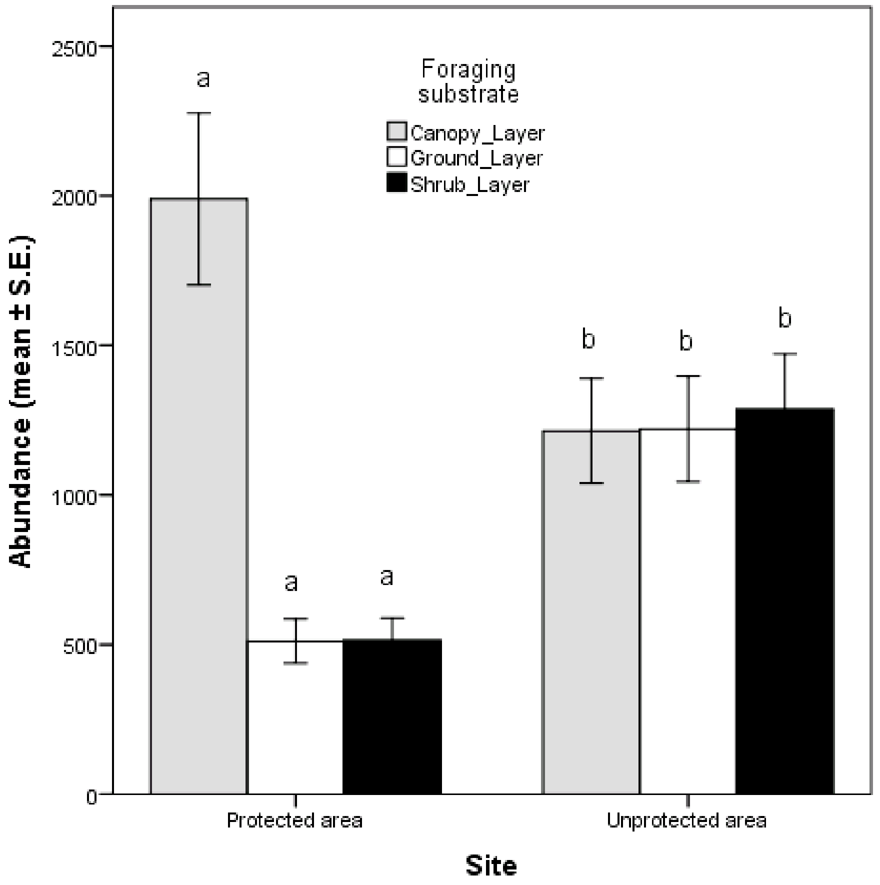
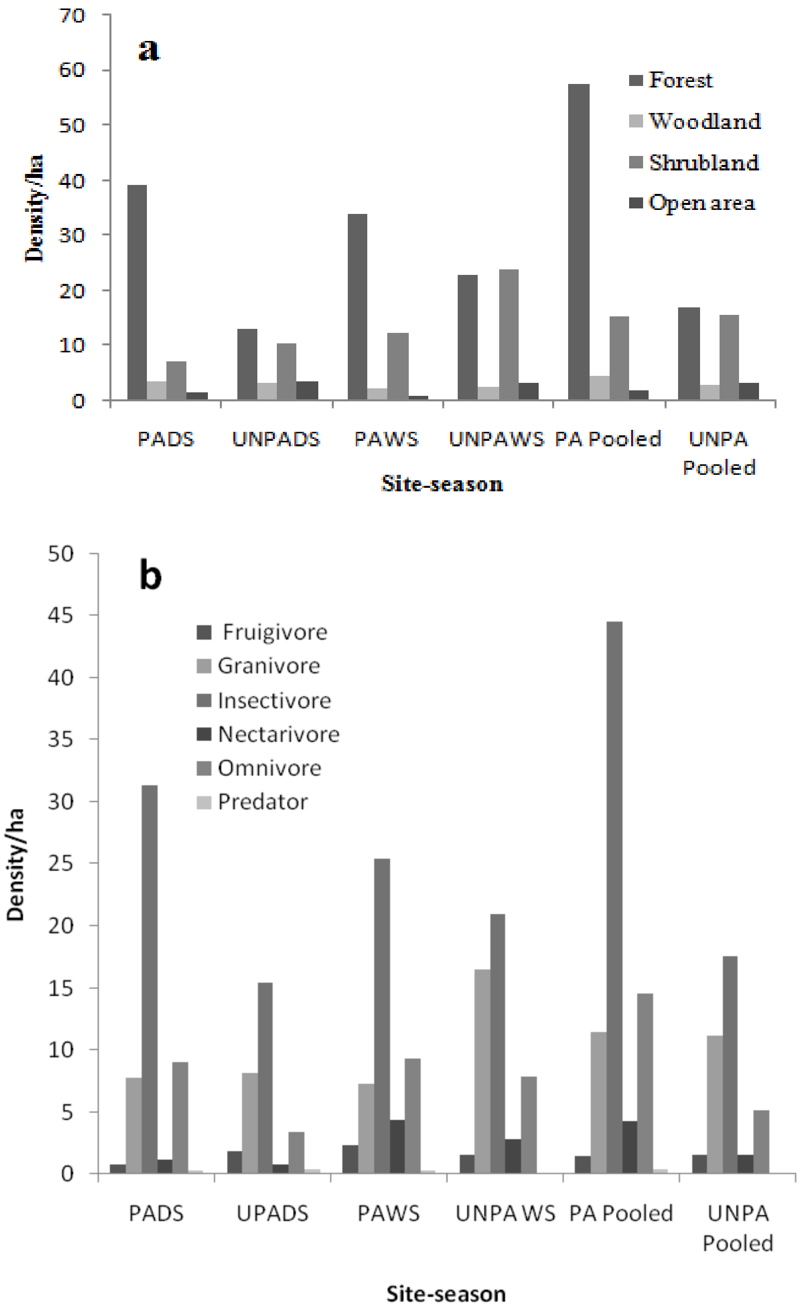


Figure 9. Mean (± 1 SE) abundance of avian feeding substrate guilds in the protected and unprotected areas for seasons combined. Similar letters in each series denote guilds which are the same statistically between sites; differences are considered significant at the α-level of 0.05.

2.4.4 Bird density

The probability detection function fitted to bird observations from the protected and unprotected areas during each season, and years pooled, showed the absence of any heaping or evasions (Appendix II Fig. a-c), indicating that the models fitted the data. Although the density of birds (individuals.ha^{-1}) was similar between the protected and unprotected areas during the wet season, the protected areas showed higher density of birds than the unprotected areas both during the dry season (protected area: 50.87 birds.ha^{-1}; unprotected area: 29.30 birds.ha^{-1}) and when seasons were pooled (protected area: 77.56 birds.ha^{-1}; unprotected area: 37.55 birds.ha^{-1}). Considering all 13 subsets of the three sets of bird guilds, the density of forest habitat guild, insectivore and omnivore feeding guilds and tree canopy layer foraging guilds were generally higher in the protected areas than the unprotected areas during both seasons. In contrast, higher densities of shrubland habitat, granivore feeding and ground layer foraging guilds were found in the unprotected areas (Fig. 10a-d). Higher than average cluster density was found in the protected areas both during the dry season (protected area, 25.92 clusters.ha^{-1}; unprotected area 14.54 clusters.ha^{-1}) and when dry and wet seasons were pooled (protected area, 40.68 clusters ha^{-1}; unprotected area, 19.70 clusters ha^{-1}), indicating that the protected areas had higher average cluster sizes (i.e., more individuals per cluster) than the unprotected areas. This curve fitted to distance sampling modelling function describes how the probability of detecting an animal decreases with increasing distance from the observer. That is, the two land use types are supposed to be subjected to spatially and temporally different detection functions measured as p (detection) values (i.e., the probability of detecting an object in a defined area) and EDR (effective detection radius) values; protected area: dry season $p = 0.19$ (EDR 19.44m), wet season ; $p = 0.19$ (EDR 19.65m) and pooled season $p = 0.19$ (EDR 19.80m); unprotected area: dry season $p = 0.18$ (EDR 19.07m), wet season $p = 0.17$ (EDR 18.25) and pooled season $p =$ 0.17 (EDR 18.39m). However, since the values between the two land use types across the seasons are very similar (see also the detection function curves depicted on Appendix ii), it is expected that the influence of these different detection probability measures on the difference observed in density of birds should be minimal.


a
Density/ha
0
10
20
30
40
50
60
70
Forest
Woodland
Shrubland
Open area
PADS
UNPADS
PAWS
UNPAWS
PA Pooled
UNPA Pooled
Site-season
b
Density/ha
0
5
10
15
20
25
30
35
40
45
50
Fruigivore
Granivore
Insectivore
Nectarivore
Omnivore
Predator
PADS
UPADS
PAWS
UNPA WS
PA Pooled
UNPA Pooled
Site-season

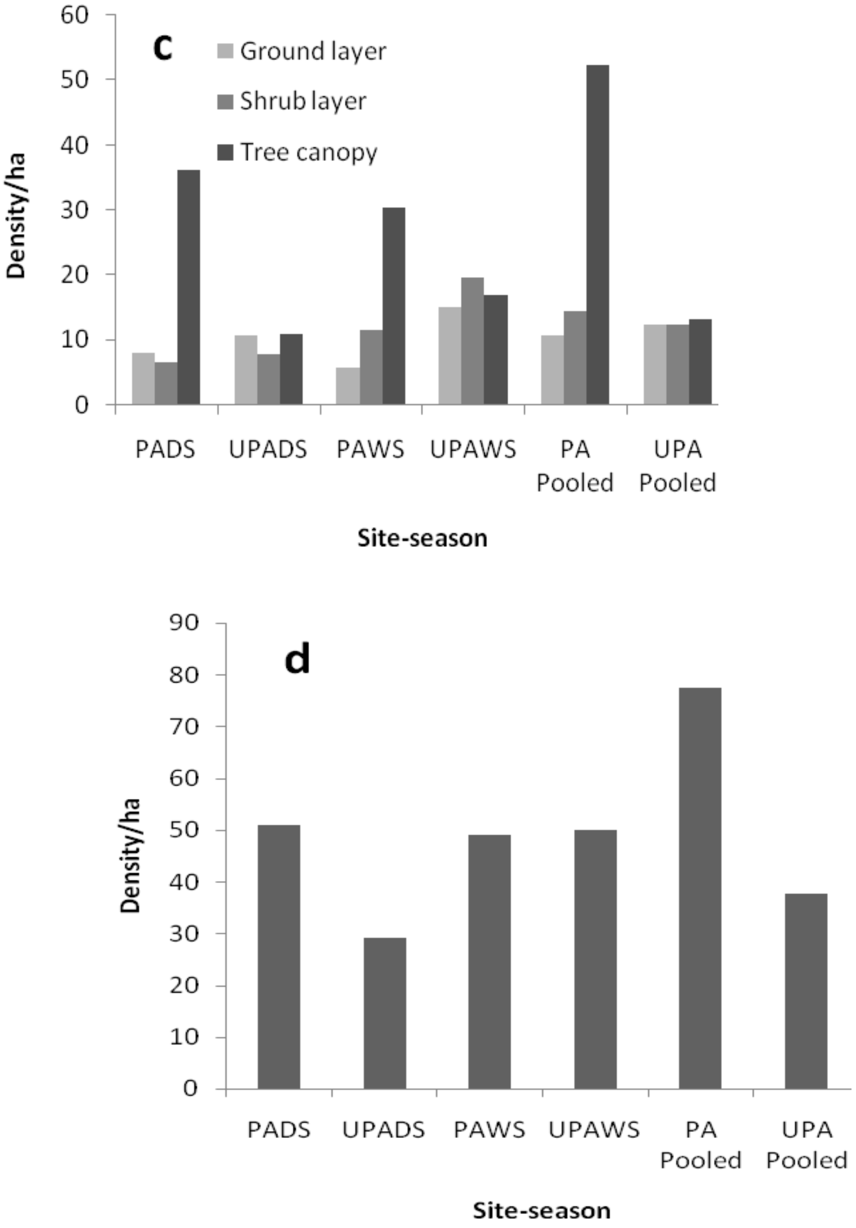


Figure 10. Estimated density (individuals. ha^{-1}; using Distance sampling) of overall and the 13 subsets of guilds of birds in the protected and unprotected areas during the dry and wet seasons and for seasons pooled (a = habitat guilds, b = feeding guilds, c = foraging substrate guilds, and d = overall species). Abbreviations: PA = protected area; UPA = unprotected area; PADS = protected area dry season; UPADS = unprotected area dry season; PAWS = protected area wet season.

2.4.5 Assemblage composition

When pair-wise tests of all possible comparisons were considered, analysis of similarity (ANOSIM) revealed differences in bird assemblages both within and between land use types across seasons, except for seasonal comparisons of the unprotected areas (Table 4). More differences were recorded between the two land use types during both seasons than seasonal differences within a land use type, differences between land use types were also more pronounced during the dry season (Fig. 11, Table 4).

Table 4. Analysis of similarity (ANOSIM) for bird assemblages within and between the two land use types during wet and dry seasons.

Site/season	**R**	**P**
Protected area DS vs Unprotected area DS	0.609	0.001
Protected area WS vs Unprotected area WS	0.461	0.001
Protected area DS vs Protected area WS	0.456	0.001
Unprotected area DS vs Unprotected area WS	**0.158**	**0.006**
Protected area pooled vs Unprotected area pooled	0.454	0.001

Bold: not significant.

Abbreviations: PADS = protected area dry season; PAWS = Protected area wet season; UPADS= unprotected area dry season; UNPAWS = unprotected area wet season.

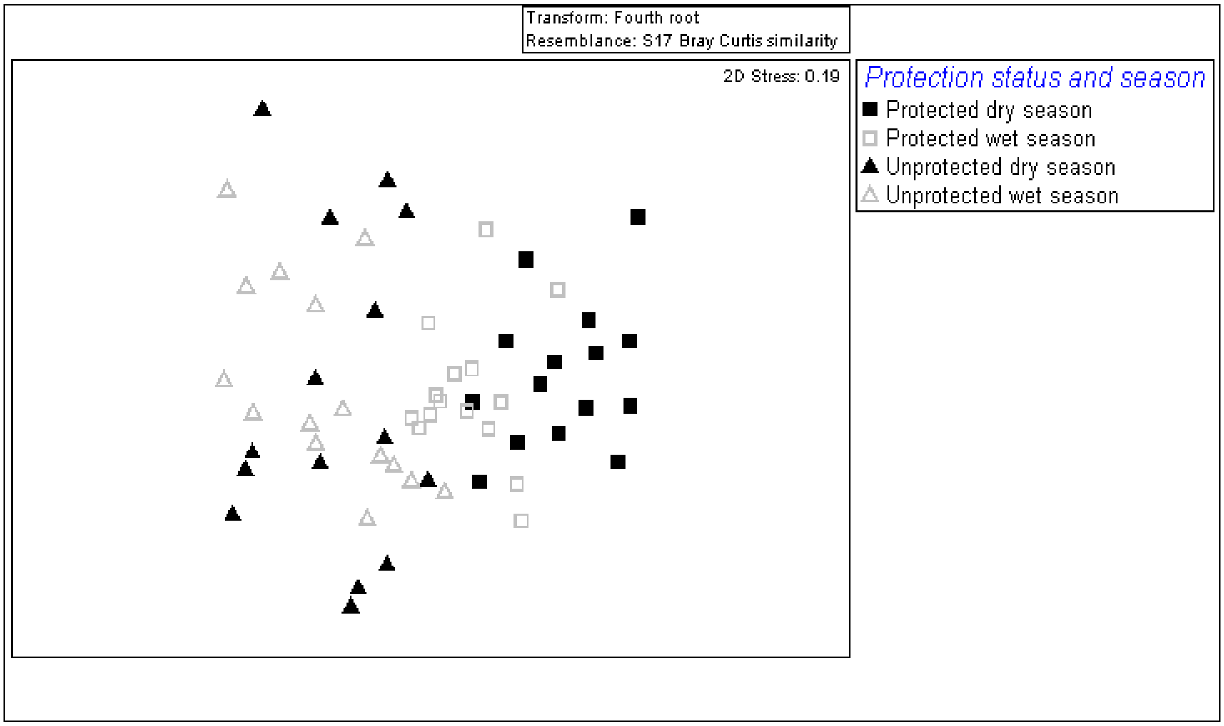


Figure 11. Non-metric multi-dimensional scaling of bird assemblage composition in protected and unprotected Afromontane forest of the BMNP across wet and dry seasons.

When considering the contribution of each guild to overall species richness and abundance, clear differences in the assemblage composition between the two land use types are revealed. For instance, forest specialists contributed 43% and 29% to overall species richness in the protected and unprotected areas, respectively. Woodland and openland species together contributed 22% and 39% to overall species in the protected areas and unprotected areas, respectively (Fig. 12 a). Relative percentage abundances of forest-specialist and shrubland species guilds were 70% and 21% in the protected areas and 42% and 40% in the unprotected areas, respectively (Fig. 13 a). Of the feeding guilds, a pronounced difference in their contribution to the overall richness and abundance was observed for granivore guild (richness

14% and 29%, abundance: 19% and 29% in the protected and unprotected areas, respectively (Figs. 12b, 13b). Further differences were revealed when foraging substrate guilds were considered; while all the three categories had almost similar percentage contribution to the overall richness and abundance in the unprotected area, the canopy layer foraging guild contributed 66% to richness.

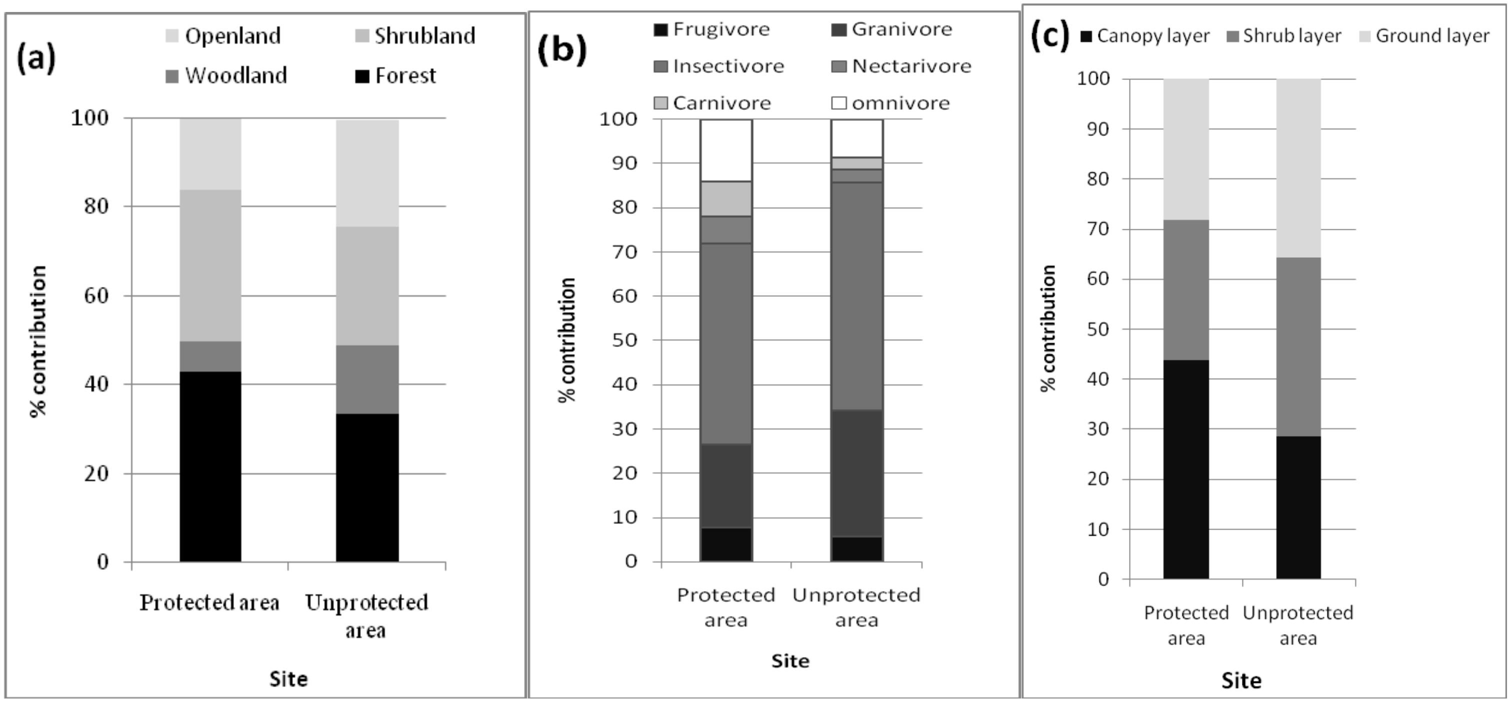


Figure 12. The relative contribution (measured as a percentage; based on combined data from both seasons) each guild of the three sets of guilds makes to the species richness of the bird assemblages in the protected and unprotected areas ((a) = habitat guilds, (b) = feeding guilds, and (c) = foraging substrate guilds).

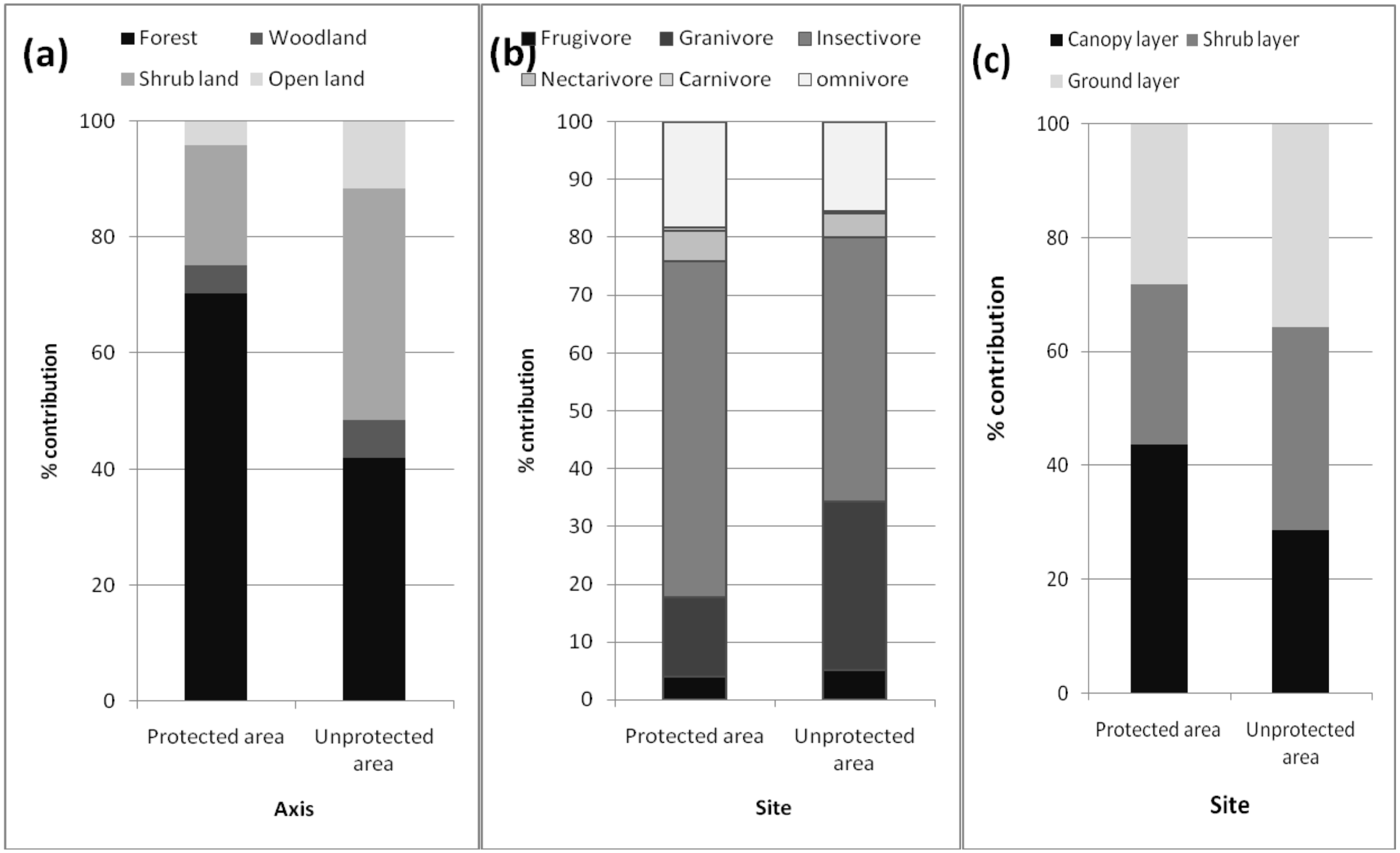


Figure 13. The relative contribution (measured as a percentage; based on combined data from both seasons) each guild of the three sets of guilds makes to the abundance of the bird assemblages in the protected and unprotected areas ((a) = habitat guilds, (b) = feeding guilds, and (c) = foraging substrate guilds)

Results of SIMPER analysis (based on pooled seasonal data) showed that 25 species (with > 1% average dissimilarity) explained 68.4% of dissimilarity to the two habitat assemblages. This included 10 forest specialists, five from openlands and 10 from shrublands. Five of the openland species and eight of shrubland species had higher average abundance in the unprotected areas than in the protected areas (Table 5). This suggests that the differences in assemblages between the two land use types is mostly due to the addition of openland and shrubland species and loss and/or decreased abundance of some forest species in the unprotected areas.

Table 5. SIMPER analysis for bird abundance (based season pooled data) between protected and unprotected areas (only those species with > 1% average dissimilarity are listed). UNAP = unprotected area, PA = protected area, Av. abund. = average abundance, Av.Diss = average dissimilarity, Cum. % = cumulative percentage of similarity.

Species	Av.Abun UNPA	Av.Abun PA	Av.Diss	Cum.%
Common Chiffchaff	0.74	1.93	1.82	3.85
Moorland Chat	1.15	0.2	1.57	7.17
Common Bulbul	1.15	0.26	1.53	10.4
Brown-rumped Seed-eater	1.35	0.64	1.53	13.63
Yellow-fronted Parrot	0	0.98	1.42	16.64
Ground Scraper Thrush	1.05	0.29	1.39	19.59
Black-winged Lovebird	1.34	0.81	1.39	22.53
Tacazze Sunbird	1.15	0.56	1.34	25.36
Abyssinian Ground Thrush	1.04	0.46	1.31	28.13
Ruppell's Robinchat	1.2	0.8	1.3	30.89
Tropical Boubou	1	0.36	1.29	33.61
Wattled Ibis	1.05	0.98	1.29	36.33

Chestnut-napped Francolin	0.57	0.99	1.28	39.03
White-cheeked Turaco	0.91	0.11	1.27	41.71
Cape Canary	0.88	0.5	1.25	44.36
Ethiopian Siskin	0.83	0.19	1.24	46.97
Tawny-flanked Prinia	1.01	0.31	1.23	49.57
Streaky Seedeater	1.52	0.92	1.2	52.11
White-backed Black Tit	1.56	2.36	1.19	54.63
Yellow-bellied Waxbill	0.73	0.21	1.14	57.04
African Dusky Flycatcher	0.81	1.27	1.11	59.39
Cinnamon-bracken warbler	1.52	1	1.11	61.72
Abyssinian Slaty Flycatcher	0.82	0.87	1.07	63.98
Cape Crow	0.71	0	1.07	66.23
Abyssinian Catbird	1.6	1.78	1.03	68.4

2.4.6 Bird–habitat relationships

Habitat variables contributing to the differences in the bird assemblages between the land use types and between seasons within each type are depicted in Table 6. Unprotected area bird assemblages showed moderately positive correlations with measures of percentage cover and height of shrubs during both seasons. Bird assemblages in the protected area also showed moderate positive correlation with shrub height. Considering the combination of habitat variables, tree abundance and shrub height showed intermediate positive correlations with the protected area assemblages. However, tree height and abundance, shrub height, % shrub cover, % forb cover and % bare ground were correlated with bird assemblages of the unprotected areas (Table 7).

Table 6. Spearman rank correlations (P_s; calculated using BioEnv procedure)) between the bird species abundance and habitat structure matrices for the protected and unprotected areas across seasons. The individual habitat variables with the three most highly correlated values for each site across seasons are shown.

Variables	Protected area dry season	Protected area wet season	Unprotected area dry season	Unprotected area wet season
Tree height	0.200	0.218	0.100	0.084
Tree abundance	0.170	0.283	-0.040	0.042
Grass height	0.067	0.091	0.031	0.080
Shrub height	-0.101	0.525	0.417	0.615
% shrub cover	-0.141	-0.077	0.547	0.512
% forbs cover	0.224	-0.004	0.410	0.106
% bare ground	0.017	0.030	0.364	0.147

Table 7. Spearman rank correlations (*Ps*; calculated using BioEnv procedure) between the bird species abundance and habitat structure matrices for the protected and unprotected areas across seasons. The combinations of habitat variables with the highest correlation for each site across seasons are shown.

Sites/seasons	Combination of habitat variables	P_r-value
Protected area/dry season	Tree height, grass height and % forbs cover	0.330
Protected area/wet season	Tree abundance and shrub height	0.549
Unprotected area/dry season	Tree height, % shrub cover, % forbs cover and % bare ground	0.604
Unprotected area/wet season	Tree abundance, shrub height, % shrub cover, grass height and bare ground	0.681

2.5 Discussion

Contrary to my predictions and despite substantial habitat disturbance in the unprotected forest patches, overall avian species richness of the unprotected Afromontane forests in the Bale Mountains of Ethiopia was on average higher than that of the protected forests. Similarly, although in agreement with my guild predictions, species grouped in woodland and open land habitat guilds, the granivore feeding guild, and the shrub and ground layer foraging substrate guilds showed significantly higher richness in the unprotected areas than in the protected areas. These results contradict the general, and perhaps more expected, trend of lower bird species richness in farmland and/or disturbed habitats compared to undisturbed forest habitats (for tropical regions, see e.g., Thiollay 1995; Daily et al. 2001; Naidoo 2004; Waltert et al. 2004; Seavy 2009; for temperate regions, see e.g., Heikkinen et al. 2004, Breitbach et al. 2010). Most studies in support of such general trends however typically investigate intensively utilized agroecosystems (e.g., Waltert et al. 2004; Seavy 2009). For instance, Waltert et al. (2004) studied natural forests compared to maize fields with little or no remaining natural vegetation, and Seavy (2009) compared birds of continuous natural forest to adjacent banana plantations. In the present study, however, the unprotected areas encompass a mosaic of agricultural-forested land cover where trees are retained in agricultural fields. The non-agricultural forested sites are impacted by selective logging and grazing, and have developed secondary growth (shrub layer) that also acts as an additional microhabitat.

Similarly to the present study, Gove et al. (2008) found equal or higher species richness both overall and per individual guilds in adjacent agricultural forestscapes than undisturbed forests in southeast Ethiopia. Furthermore, Mengesha et al. (2011; central Rift valley, Ethiopia) and Mulwa et al. (2012; western Kenya) found higher bird species richness in disturbed forests than in relatively intact adjacent forests. For most of the species (in all guilds including the insectivores) associated with the disturbed habitats in their studies, Gove et al. (2013) and Mulwa et al. (2012) hypothesised that there is connectivity both in time and space to other open areas, including those cultivated for agriculture for long periods. There are thus many birds in the species pool that have evolved in more open areas (savannas in lowlands, alpine areas in the highlands, or wetlands) and have since adapted to variegated agricultural areas and can colonize forest areas that have been disturbed and converted. Another explanation could be that African bird species, including sensitive guilds such as understory insectivores may not be as sensitive to forest conversion compared to species found elsewhere due to several thousand years of forest clearance and agrarian activity in Africa (Chapman & Chapman 1996; Darbyshire et al. 2003), causing African fauna to be more tolerant to disturbance (Karr 1976; Gove et al. 2008; Mulwa et al. 2012; Gove et al. 2013).

In addition to these possibilities, more specific explanations can be made regarding the higher species richness in the unprotected area relative to the protected area in the present study. Previous studies have pointed out that land-use intensity and thus the structural diversity in tropical farmlands strongly influences bird diversity (Harvey et al. 2006; Şekercioğlu et al. 2007; Laube et al. 2008). Thus, the higher bird species richness reported in the present study for the unprotected areas may partly be attributable to their higher structural diversity, which comprises both primary and secondary forest patches, forest galleries, openland areas and crops with retained canopy trees. These heterogeneous structural elements may constitute different micro-habitats and niches for a wider variety of bird species compared to the less disturbed forests that are typically dominated by primary growth with less understory growth (Tews et al. 2004; Mengesha et al. 2011). Although it is understood that crop fields attract granivore guilds (see

Mulwa et al. 2012; Gove et al. 2013), the reason for higher numbers of insectivore species in the unprotected area is not clear, especially since sub-guilds (e.g. understory insectivores) of these species are known to represent the guild that is the most sensitive to structural changes in forest vegetation (Canaday 1996; Şekercioğlu 2002; Tscharntke et al. 2008). A possible lack of farming inputs (such as insecticides) in the area, and therefore little impact on insect abundance, could have contributed to the persistence of insectivore birds in the unprotected areas. A lack of significant differences in species richness of frugivores and omnivores between the two land use types is consistent with previous studies (e.g. Waltert et al. 2004; Gomes et al. 2008). These authors have suggested that these groups of species are less habitat-specific as long as food sources are available to them and thus are more tolerant to habitat alteration. Omnivorous species, particularly, have a wide range of food sources making them well suited to modified areas.

Most of the guilds for which higher species richness in the unprotected areas was reported, also demonstrated higher abundances and densities (individuals ha^{-1}) in unprotected compared to protected areas. The role of predation in shaping bird density between the land use types was also minimal since the species richness and density of raptors is similar in both. Therefore, ecosystem processes such as resource (e.g. food and habitat) availability and inter- and/or intra-specific interactions are likely the underlying factors for the differences in density of birds detected between the two land use types examined. Such a resource gradient, in turn, could be explained as a consequence of differences in disturbance levels between the two land use types. Thus the presence of crop fields, openland areas and disturbance-induced shrub development in the unprotected areas could have created favourable conditions for such non-forest specialist species.

The species composition of bird communities differed substantially between the protected and unprotected areas; the latter being dominated by (i) woodland and open land habitat guilds, (ii) granivore and insectivore feeding guilds, and (iii) shrub layer and ground layer foraging substrate guilds. In contrast, forest habitat and canopy layer foraging species guilds dominate the assemblage in the protected areas. This difference was more pronounced during the dry season, when the difference in habitat structural composition was also more pronounced between the two land use types, suggesting that differences in habitat structure is a principal factor contributing to the differences observed in assemblage composition (Rahayuningsih et al. 2007; Mengesha et al. 2011). This season is the time when crops are harvested, attracting openland ground layer foraging species while having negative impacts on those species preferring shrub habitat, hence enhancing the compositional differences between the two habitats.

2.6 Conclusion

The result that current disturbance levels found in the Afromontane forests of the Bale Mountains lead to an increase in the overall bird species richness compared to less disturbed and protected forests should be interpreted with caution. Indeed, this is because bird densities for all species examined, forest-specialists and those species that forage in the tree canopy were higher in the protected areas compared to unprotected areas. Similarly, and in addition, the relative contribution of each guild to the overall richness and abundances of the assemblages showed that the forest-specialists and those species foraging in the canopy layer were more pronounced in the protected areas; and conversely, woodland-, shrubland- and openland-specialists together with granivores were dominant in the unprotected areas. This suggests that forest-specialist species are negatively affected by habitat disturbance, with non-forest-specialist species favored.

From a park management, the objective of protecting the Afromontane forest of the Bale Mountains from an avian perspective should remain that of maximising all aspects of the overall bird diversity associated with this habitat type (i.e. forest species, see OBARD 2007). The implications of the present findings, namely that disturbance causes encroachment of non-native species (openland, woodland and shrub land species), and negatively affects the density and abundance of native forest species, particularly tree canopy foragers, is that park management and decision making efforts should focus on how best to avoid or minimize forest degradation within the protected areas in order to maximize the overall diversity of bird species that are native and specialist to these biological important forests with global significance. If effective conservation efforts are not implemented and enforced in those forests within protected areas, and the rate of forest destruction in the Bale Mountains continues at the current trend (see Teshome et al. 2011), the local extinction of forest-specialists is inevitable. Similarly, Stephens et al. (2001) showed that settlement and agricultural expansion in the northern Bale Mountains also significantly threatens larger mammal species of the region. Future work should therefore focus on comparing several sites with varying levels of disturbances (e.g. none, low, medium, heavy, and very heavy) in order to better understand the impacts of disturbance on biodiversity, in general, and birds in particular.

2.7 References

Abera, K., and A.A. Kinahan. 2011. Factors Affecting Fire Extent and Frequency in the Bale Mountains National Park. Walia-Special Edition on the Bale Mountains 146-157.

African Bird Club. 2012. http://www.africanbirdclub.org/countries/checklists/index.html. Last update on 3 August 2012 (accessed on October 2012).

Asefa, A. 2006-07. Birds of Bale Mountains National Park, Southeast Ethiopia. Walia **25**: 22-33.

Asefa, A. 2011. Mammals of the Bale Mountains National Park, Ethiopia: a compiled and annotated checklist. Walia-Special Edition on the Bale Mountains 3-14.

Bibby, C., M. Jones, and S. Marsden. 1998. Expedition field techniques: bird surveys. The expedition Advisory Centre, Royal Geographical Society. The institute of British geographers, London, United Kingdom.

Bleher, B., D. Uster, and T. Bergsdorf. 2006. Assessment of threat status and management effectiveness in Kakamega Forest, Kenya. Biodiversity and Conservation **15**:1159-1177.

Breitbach, N., I. Laube, I. Stefan-Dewenter, and K. Böhning-Gaese. 2010. Bird diversity and seed dispersal along a human land-use gradient: High seed removal in structurally simple farmland. Oecologia **162**: 965–976.

Brooks, T.M., R. A. Mittermeier, G. A. B. da Fonseca, J. Gerlach, M. Hoffmann, J. F. Lamoreux, C. G. Mittermeier, J. D. Pilgrim, and A. S. L. Rodrigues. 2006. Global biodiversity conservation priorities. Science **313**: 58-63.

Bruner, A.G., R.E. Gullison, R.E. Rice, and G.A.B. da Fonseca. 2001. Effectiveness of Parks in Protecting Tropical Biodiversity. Science **291**:125-127.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to distance sampling. Oxford University Press, Oxford, United Kingdom.

Canaday, C. 1997. Loss of insectivorous birds along a gradient of human impact in Amazonia. Biological Conservation **77**: 63-77.

Chao, A. 2004. Species richness estimation. In Encyclopedia of statistical sciences. Vol. 12: 7907–7916. Balakrishnan, N., Read, C.B. & Vidakovic, B. (Eds). Wiley Press, New York, USA.

Chapman C.A., and L.J. Chapman. 1996. Mid-elevation forests: A history of disturbance and regeneration. Pages 385-400 in T.R. McClanahan, and T.P. Young, editors. East African Ecosystems and Their Conservation. Oxford University Press, Oxford, United Kingdom.

Chown, S.L. 2010. Temporal biodiversity change in transformed landscapes: a southern African perspective. Philosophical Transactions of the Royal Society, Biological sciences **365**: 3729-3742.

Clarke, K. R. 1993. Non-parametric multivariate analyses of change in community structure. Australian Journal of Ecology **18**:117-143.

Clarke, K. R., and R. M. Warwick. 1994. Change in marine communities: an approach to statistical analysis and interpretation. Plymouth Marine Laboratory, Plymouth, United Kingdom.

Clarke, K.R., and R.N. Gorley. 2006. PRIMER V6: User Manual/Tutorial. PRIMER-E, Plymouth, United Kingdom.

Coleman, B.D., 1981. On random placement and species-area relations. Mathematical Bioscience **54**:191–215.

Colwell, R. K. 2009. EstimateS 8.2 User's Guide. Website: http://purl.oclc.org/estimates or http://viceroy.eeb.uconn.edu/estimates.

Darbyshire I., H. Lamb, and M. Umer. 2003 . Forest clearance and regrowth in northern Ethiopia during the last 3000 years. The Holocene **13**: 537-546.

Daily, G. C., P. R. Ehrlich, and G. A. Sanchez-Azofeifa. 2001. Countryside biogeography: Use of human-dominated habitats by the avifauna of southern Costa Rica. Ecological Applications **11**:1–13.

EWNHS. 1996. Important bird areas of Ethiopia: A first inventory**.** Ethiopian Wildlife and Natural History Society, Addis Ababa, Ethiopia

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology Evolution and Systematics **34**: 487-515.

Game, E.T., Kareiva, P. and Possingham, H.P. 2013. Six Common Mistakes in Conservation Priority Setting. Conservation Biology DOI: 10.1111/cobi.12051.

Gaston, K. J., S.F. Jackson, L. Cantu-Salazar, and G. Cruz-Pinon. 2008. The Ecological Performance of Protected Areas. Annual Review of Ecology Evolution and Systematics **39**: 93-113.

Gibbons, D.W., D.A. Hill, and J.W. Sutherland. 1996. Birds. Pages 227-259 in W.J. Sutherland editor. Ecological census techniques: A hand book. Cambridge University Press, Cambridge, United Kingdom.

Gomes, L. G. L., V. Oostra, V. Nijman, A. M. Cleef, and M. Kappelle. 2008. Tolerance of frugivorous birds to habitat disturbance in a tropical cloud forest. Biological Conservation **141**: 860–871.

Gotelli, N., and R. K. Colwell. 2001. Quantifying biodiversity: Procedures and pitfalls in the measurement and comparison of species richness. Ecology Letters **4**: 379-391.

Gove, A.D., K. Hylander, S. Nemomisa, and A. Shimelis. 2008. Ethiopian coffee cultivation-Implications for bird conservation and environmental certification. Conservation Letters **1**: 208-216.

Gove, A.D., K. Hylander, S. Nemomsa, A. Shimelis, and W. Enkossa. 2013. Structurally complex farms support high avian functional diversity in tropical montane Ethiopia. Journal of Tropical Ecology **29**: 87-97.

Gregory, R.d., D.W. Gibbons, and P.F. Donald. 2004. Bird census and survey techniques. Pages 17-56 in W.J. Sutherland, I. Newton, and R. Green, editors. Bird ecology and conservation: a handbook of techniques -Issue 1 of techniques in ecology and conservation series. Oxford University Press, Oxford, United Kingdom.

Harvey, C. A., A. Medina, D. M. Sánchez, S. Vílchez, B. Hernández, J. C. Saenz, J. M. Maes, F. Casanoves, and F. L. Sinclair. 2006. Patterns of animal diversity in different forms of tree cover in agricultural landscapes. Ecological Applications **16**: 1986–1999.

Heikkine, R., M. Luoto, R. Virkkala, and K. Rainio. 2004. Effects of habitat cover, landscape structure and spatial variables on the abundance of birds in agricultural-forest mosaic. Journal of Applied Ecology **41**: 824-835.

Hillman, J.C. 1986. Bale Mountains National Park Management Plan. Ethiopian Wildlife Conservation organization, Addis Ababa, Ethiopia.

IBM Corporation. 2011. IBM SPSS Statistics 20, version 20. IBM Corporation, USA.

Jackson, S.F., K. Walker, and K.J. Gaston. 2009. Relationship between distributions of threatened plants and protected areas in Britain. Biological Conservation **142**:1515-1522.

Jensen, A. 1985. The effect of cattle and sheep grazing on salt marsh vegetation at Skallingen, Denmark. Vegetatio **60**: 37-48.

Karr J.R. 1976. Within- and Between-Habitat Avian Diversity in African and Neotropical Lowland Habitats. Ecological Monographs **46**: 457-481.

Kerley, G.H., and W.G. Whitford. 2000. Impact of Grazing and Desertification in the Chihuahuan Desert: Plant Communities, Granivores and Granivory. American Midland Naturalist **144**: 78-91.

Kessler, M., P.J.A. Kebler, S.R. Grandstein, K. Bach, M. Schmill, and R. Pitopang. 2005. Tree diversity in primary forest and different land use systems in central Sulawes, Indonesia. Biodiversity and Conservation **14**: 547-560.

Kissling W. D., C.H., Şekercioğlu, and J. Walter. 2011. Bird dietary guild richness across latitudes, environments and biogeographic regions. Global Ecology and Biogeography1-13.

Kumar, A., and J. Ram. 2005. Anthropogenic disturbances and plant biodiversity in forests of Uttaranchal, Central Himalaya. Biodiversity and Conservation **14**: 309-331.

Largen, M., and S. Spawls. 2011. Amphibians and Reptiles Recorded from the Bale Mountains. Walia-Special Edition on the Bale Mountains 89-91.

Laube, I., N. Breitbach, and K. Böhning-Gaese. 2008. Avian diversity in a Kenyan agroecosystem: Effects of habitat structure and proximity to forest. Journal of Ornithology **149**:181–191.

Locke, H., and P. Dearden. 2005. Rethinking protected area categories and the new paradigm. Environmental Conservation **32**:1-10.

Longino, J.T., J. Coddington, and R.K. Colwell. 2002. The Ant Fauna of a Tropical Rain Forest: Estimating Species Richness Three Different Ways. Ecology **83**: 689-702.

MacArthur, R. H, and J. W. MacArthur. 1961. On bird species diversity. Ecology **42**: 594–598.

Magurran, A. E. 2004. Measuring biological diversity. Blackwell, Oxford, United Kingdom.

Martin, T.E. 1984. Impact of livestock grazing on birds of a Colombia Cloud Forest. Tropical Ecology. **25**:158-171.

Martin, T.G., and H.P. Possingham. 2005. Predicting the impact of livestock grazing on birds using foraging height data. Journal Applied Ecology **42**: 400-408.

Mengesha, G., Y. Mamo, and A. Bekele. 2011. A comparison of terrestrial bird community structure in the undisturbed and disturbed areas of the Abijata Shalla lakes national park, Ethiopia. International Journal of Biodiversity and Conservation **3**: 389-404.

Miehe, G., and S. Miehe. 1994. Ericaceous forest and heath land in Bale Mountains of South Ethiopia, Ecology and Man's Impact. Traut Warnke Verlag, Hamburg, Germany.

Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: biodiversity synthesis. World Resources Institute, Washington, D.C.

Mulwa, R.K., K. Böhning-Gaese, and M. Schleuning. 2012. High Bird Species Diversity in Structurally Heterogeneous Farmland in Western Kenya. Biotropica **44**:1–9.

Naidoo, R. 2004. Species richness and community composition of song birds in a tropical forest-agricultural landscape. Animal Conservation **7**: 93–105.

Newton, A.C. 2007. Forest ecology and conservation: a handbook of techniques in ecology and conservation series. W.J. Sutherland series editor. Oxford University Press, Oxford, United Kingdom.

NH (The National Herbarium). 2004. Biodiversity assessment of the Bale Mountains National Park and surrounding areas. The National Herbarium of Addis Ababa University, Addis Ababa, Ethiopia.

OARDB. 2007. Bale Mountains National Park General Management Plan. Oromia Agriculture and Rural Development Bureau, Finfinnee, Ethiopia.

Rahayuninagsih, M., A. Mardiastuti, L. Prasetyo, and Y. Mulyani. 2007. Bird community in Burung Island, Karimunjawa National Park, Central Java. Biodiversity **8**:183-187.

Recher, H.F. 1969. Bird species diversity in Australia and North America. The American Naturalist **103**: 75-80.

Redman, N., T. Stevenson, and J. Fanshawe. 2009. Birds of the Horn of Africa: Ethiopia, Eritrea, Djibouti, Somalia and Socotra. Christopher Helm, London.

Seavy, N. E. 2009. Bird use of banana plantations adjacent to Kibale national park, Uganda: Evaluating the conservation value of a matrix habitat. Journal of East African Natural History **98**: 211–222.

Şekercioğlu, C.H. 2002. Effects of forestry practices on vegetation structure and bird community of Kibale National Park, Uganda. Biological Conservation **107**: 229-240.

Şekercioğlu, C. H., P.R. Ehrlich, G. C. Daily, D. Aygen, D. Goehring, and R.F. Sandí. 2002. Disappearance of insectivorous birds from tropical forest fragments. Proceedings of the National Academy of Science of USA **99**: 263-267.

Shimelis, A., A. Bekele, A. Asefa , S. Wiliams, A. Gove, and S. Thirgood. 2011. Structuring of the Birds of the Bale Mountains National Park. Walia-Special Edition on the Bale Mountains 15-27.

Sinclair, A. R. E., S.A.R. Mdluma, and P. Arcese. 2002. Protected areas as biodiversity benchmarks for human impact: agriculture and the Serengeti avifauna. Proceedings of Royal Society of London Series- B Biological Sciences 269: 2401-2405.

Sinclair, I. and P. Ryan. 2003. Birds of Africa south of the Sahara: A comprehensive illustrated field guide. Struik publishers, Cape Town, South Africa.

Stephens, P.A., C.A. D'sa, C. Sillero-Zubri, and N.L. Williams. 2001. Impact of livestock and settlement on the large mammalian wildlife of Bale Mountains National Park, Southern Ethiopia. Biological Conservation **100**: 307-322.

Struhsaker, T.T., P.J. Struhsaker, and K.S. Siex. 2005. Conserving Africa's rain forests: problems in protected areas and possible solutions. Biological Conservation **123**: 45-54.

Tabeni, S. and R.A. Ojeda. 2005. Ecology of the Monte Desert small mammals in disturbed and undisturbed habitats. Journal of Arid Environments **63**: 244-255

Teshome, E., D. Randal, and A.A. Kinahan. 2011. The Changing Face of the Bale Mountains National Park over 32 years: A Study of Land Cover Change. Walia-Special Edition on the Bale Mountains 118-130.

Tews, J., U. Brose, V. Grimm, K. Tielbörger, M. C. Wichmann, M. Schwager, and F. Jeltsch. 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. Journal of Biogeography **31**:79-92.

Thiollay, J. M. 1995. The role of traditional agroforests in the conservation of rain forest bird diversity in Sumatra. Conservation Biology **9**: 335–353.

Thomas, L., J.L. Laake, S. Strindberg, F.F.C. Marques, S.T. Buckland, D.L. Borchers, D.R. Anderson, K.P. Burnham, S.L. Hedley, J.H. Pollard, and J.R.B. Bishop. 2003. Distance 4.1. Release 2. Research Unit for Wildlife Population Assessment, University of St. Andrews, UK. http://www.ruwpa.stand. ac.uk/distance/.

.

Trscinski, M.K., L. Fahrig, and G. Merriam. 1999. Independent effects of forest cover and fragmentation on the distribution of forest breeding birds. Ecological applications **9**:586-593.

Tscharntke T., C.H. Şekercioğlu, T.V. Dietsch, N.S. Sodhi, P. Hoehn, and J.M. Tylianakis. 2008. Landscape constraints on functional diversity of birds and insects in tropical agroecosystems. Ecology **89**: 944-951.

Urban, E. K., and L. E. Brown. 1971. A checklist of the birds of Ethiopia. Haile Sellassie I University Press, Addis Ababa, Ethiopia.

van Rensburg, B.J., S.L. Chown, A.S. van Jaarsveld, and M.A. McGeoch. 2000. Spatial variation and biogeography of sand forest avian assemblages in South Africa. Journal of Biogeography **27**: 1385-1401.

van Rensburg, B.J., D.S. Peacock, and M.P. Robertson. 2009. Biotic homogenization and alien bird species along an urban gradient in South Africa. Landscape and Urban Planning **92**: 233-241.

van Schaik C.P., J. Terborgh, and D. Dugelby. 1997. The silent crisis: the state of the rainforest nature reserves. Pages 64-89 in R. Kramer, C.P. van Schaik, and J. Johnson, editors. The Last Stand: Protected Areas and the Defense of Tropical Biodiversity. Oxford University Press, Oxford, United Kingdom.

Vickery, J. A., J. R. Tallowin, R. E. Feber, E. J. Asteraki, P. W. Atkinson, R. J. Fuller, and V. K. Brown. 2001. The Management of Lowland Neutral Grasslands in Britain: Effects of Agricultural Practices on Birds and Their Food Resources. Journal of Applied Ecology **38**: 647-664.

Waltert, M., A. Mardiastuti, and M. Mühlenberg. 2004. Effects of land use on bird species richness in Sulawesi, Indonesia. Conservation Biology **18)**:1339–1346.

Woldu, Z., and M.A. Mohammed Saleem. 2000. Grazing induced biodiversity in the highland ecozone of East Africa. Agriculture, Ecosystems and Environment **79**: 43-52.

Williams, S., J.L. Vivero Pol, S. Spawls, A. Shimelis, and E. Kelbessa. 2004. Ethiopian Highlands. Pages 262-273 in (R.A. Mittermeier, P.R. Gill, M. Hoffmann, J. Pilgrim, T. Brooks, C.G. Mittermeier, J. Lamoreux, and G.A.B. Da Fonseca editors. Hotspots revisited. CEMEX Publisher, Mexico City, Washington, D.C.

Wynne, G. 1998. Conservation policy and politics. Pages 256-285 in W.J. Sutherland editor. Conservation, Science and Action. Blackwell Science, Oxford, United Kingdom.

Yalden, D.W. 1983. The extent of high ground in Ethiopia compared to the rest of Africa. SINET: Ethiopian Journal of Sciences **6**: 256-285.

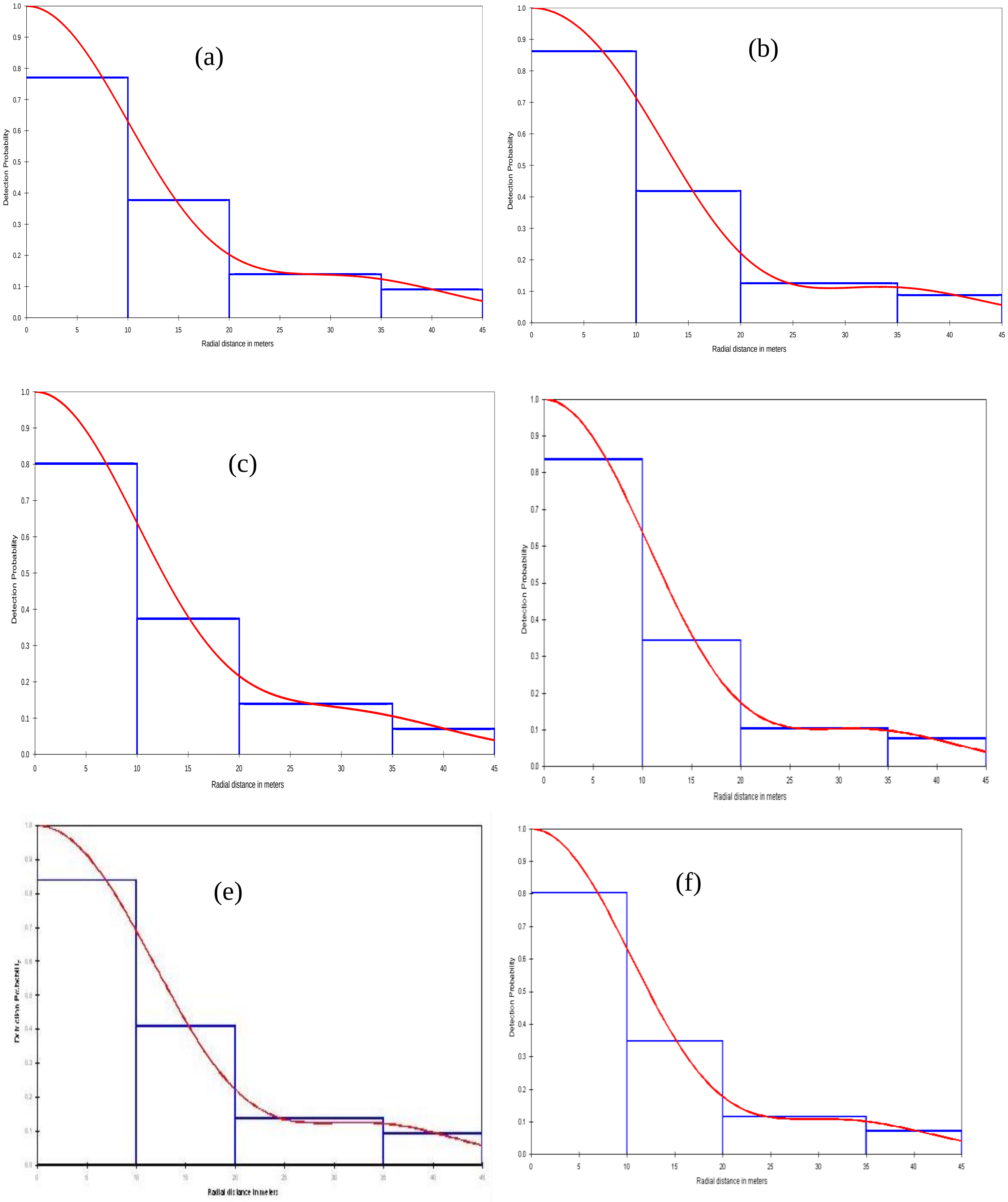
Appendix I. List of species recorded in the unprotected and protected areas of the Afromontane forests in the northern Bale Mountains, Ethiopia (numbers under the land use type columns refer to the overall abundance of each species). Nomenclature follows the checklist of African Bird Club (2012).

Common name	Scientific name	Habitat ID	Food type ID	Foraging Substrate ID	Unprotected area	Protected area
Abyssinian Catbird	*Parophasma galinieri*	Forest	Insectivore	Tree canopy	558	420
Abyssinian Crimsonwing	*Cryptospiza salvadorii*	Forest	Granivore	Shrub layer	4	
Abyssinian Ground Hornbill	*Bucorvus abyssinicus*	Openland	Insectivore	Ground layer	2	
Abyssinian Ground Thrush	*Zoothera piaggiae*	Forest	Insectivore	Ground layer	80	48
Abyssinian Slaty Flycatcher	*Melaenornis chocolatinus*	Woodland	Insectivore	Tree canopy	55	93
Abyssinian Woodpecker	*Dendropicos abyssinicus*	Forest	Insectivore	Tree canopy	16	39
African Citril	*Serinus citrinelloides*	Shrubland	Granivore	Shrub layer	8	2
African Dusky Flycatcher	*Muscicapa adusta*	Forest	Insectivore	Tree canopy	61	142
African Emerald Cuckoo	*Chrysococcyx cupreus*	Shrubland	Insectivore	Shrub layer	1	
African Goshawk	*Accipiter tachiro*	Forest	Carnivore	Tree canopy	2	5
African Olive Pigeon	*Columba arquatrix*	Forest	Fruigivore	Tree canopy	47	
Baglafecht Weaver	*Ploceus baglafecht*	Shrubland	Insectivore	Shrub layer	78	9
Black-winged Lovebird	*Agapornis taranta*	Forest	Fruigivore	Tree canopy	171	157
Brown Woodland Warbler	*Phylloscopus umbrovirens*	Forest	Insectivore	Tree canopy	390	684
Brown-rumped Seedeater	*Serinus tristriatus*	Shrubland	Granivore	Ground layer	277	66

Cape Canary	*Serinus canicollis*	Shrubland	Granivore	Shrub layer	440	291
Cape Crow	*Corvus capensis*	Openland	Ominivore	Ground layer	50	
Chestnut-naped Francolin	*Francolinus castaneicollis*	Shrubland	Ominivore	Ground layer	106	141
Cinnamon Bracken Warbler	*Bradypterus cinnamomeus*	Shrubland	Insectivore	Shrub layer	234	98
Collared Sunbird	*Hedydipna collaris*	Shrubland	Nectarivore	Tree canopy	18	
Common Bulbul	*Pycnonotus barbatus*	Shrubland	Ominivore	Shrub layer	228	9
Common Chiffchaff	*Phylloscopus collybita*	Forest	Insectivore	Tree canopy	22	251
Common Fiscal	*Lanius collaris*	Woodland	Carnivore	Shrub layer	15	
Common Waxbill	*Estrilda astrild*	Shrubland	Granivore	Shrub layer	125	10
Dusky Turtle Dove	*Streptopelia lugens*	Woodland	Granivore	Ground layer	336	184
Ethiopian Siskin	*Serinus nigriceps*	Openland	Granivore	Ground layer	155	16
Fan-tailed Raven	*Corvus rhipidurus*	Openland	Ominivore	Ground layer	42	24
Grey Wagtail	*Motacilla cinerea*	Openland	Insectivore	Ground layer	3	
Grey-headed Woodpecker	*Dendropicos spodocephalus*	Forest	Insectivore	Tree canopy	1	13
Ground Scraper Thrush	*Psophocichla litsitsirupa*	Openland	Insectivore	Ground layer	95	18
Hoopoe	*Upupa epops*	Woodland	Insectivore	Ground layer	17	
Little Sparrowhawk	*Accipiter minullus*	Forest	Carnivore	Tree canopy		1
Malachite Sunbird	*Nectarinia famosa*	Shrubland	Nectarivore	Shrub layer		1
Montane White-eye	*Zosterops poliogastrus*	Forest	Ominivore	Tree canopy	406	824
Moorland Chat	*Cercomela sordida*	Openland	Insectivore	Ground layer	178	12

Narina's Trogon	*Apaloderma narina*	Forest	Insectivore	Tree canopy		1
Northern Puffback	*Dryoscopus gambensis*	Forest	Insectivore	Tree canopy	6	
Nubian Woodpecker	*Campethera nubica*	Forest	Insectivore	Tree canopy		2
Olive Thrush	*Turdus olivaceus*	Forest	Insectivore	Ground layer	462	203
Pied Crow	*Corvus albus*	Openland	Ominivore	Ground layer	4	
Pin-tailed Whydah	*Vidua macroura*	Shrubland	Granivore	Shrub layer	2	
Red-collared Widowbird	*Euplectes ardens*	Shrubland	Granivore	Shrub layer	26	
Red-eyed Dove	*Streptopelia semitorquata*	Woodland	Granivore	Ground layer	10	
Red-throated Pipit	*Anthus cervinus*	Openland	Insectivore	Ground layer	50	
Red-throated Wryneck	*Jynx ruficollis*	Woodland	Insectivore	Tree canopy	5	1
Rouget's Rail	*Rougetius rougetii*	Shrubland	Ominivore	Ground layer		2
Rufous-chested Sparrowhawk	*Accipiter rufiventris*	Forest	Carnivore	Tree canopy	4	22
Rüppell's Robin-Chat	*Cossypha semirufa*	Forest	Insectivore	Ground layer	195	84
Slender-billed Starling	*Onychognathus tenuirostris*	Shrubland	Ominivore	Shrub layer	235	10
Speckled Mousebird	*Colius striatus*	Woodland	Fruigivore	Tree canopy	10	
Speckled Pigeon	*Columba guinea*	Openland	Granivore	Ground layer	12	
Streaky Seedeater	*Serinus striolatus*	Shrubland	Granivore	Shrub layer	408	135
Swainson's Sparrow	*Passer swainsonii*	Openland	Granivore	Ground layer	50	
Tacazze Sunbird	*Nectarinia tacazze*	Shrubland	Nectarivore	Shrub layer	274	299
Tawny-flanked Prinia	*Prinia subflava*	Shrubland	Insectivore	Shrub layer	120	30

Thekla Lark	*Galerida theklae*	Openland	Insectivore	Ground layer	24	4
Thick-billed Raven	*Corvus crassirostris*	Openland	Ominivore	Ground layer	11	22
Tropical Boubou	*Laniarius aethiopicus*	Forest	Insectivore	Tree canopy	104	33
Wattled Ibis	*Bostrychia carunculata*	Openland	Insectivore	Ground layer	102	134
White-backed Black Tit	*Parus leuconotus*	Forest	Insectivore	Tree canopy	275	968
White-cheeked Turaco	*Tauraco leucotis*	Forest	Fruigivore	Tree canopy	125	14
Winding Cisticola	*Cisticola galactotes*	Shrubland	Insectivore	Shrub layer	30	3
Yellow Bishop	*Euplectes capensis*	Shrubland	Granivore	Shrub layer	9	
Yellow Wagtail	*Motacilla flava*	Openland	Insectivore	Ground layer	26	
Yellow-bellied Waxbill	*Estrilda quartinia*	Shrubland	Granivore	Shrub layer	174	67
Yellow-fronted Parrot	*Poicephalus flavifrons*	Forest	Fruigivore	Tree canopy		59



Appendix II. Probability detection function fitted to bird observations from the protected and unprotected areas of Afromontane forest in the Bale Mountains for (a) protected areas during the dry season, (b) protected areas during the wet season, (c), unprotected areas during the dry season (d), unprotected areas during the wet season, (e)) protected areas, seasons pooled and (f) unprotected areas, seasons pooled.

Chapter 3

Birds as bioindicators for forest change:

a case study from the Bale Mountains of Ethiopia

Abstract: *As a result of increased anthropogenic threats, poor management systems and limited finances, many protected areas in tropical developing countries are in danger of losing their biodiversity and ecological value. Thus, identifying early warning systems for habitat change is a critical measure in helping to mitigate the effects of these threats. This study was undertaken to examine the usefulness and applicability of birds as bioindicators in dry Afromontane forests in the northern Bale Mountains of southeast Ethiopia, and their potential application for forest monitoring and conservation. Birds were surveyed in three protected and three unprotected forest patches in 2009 and 2012 using a point-transect method. The IndVal procedure was used to identify potential indicator species (species characteristic of a particular habitat) and detector species (species with some degree of habitat preference); this was refined by the use of other species-specific criteria such as baseline information, location information, and niche and life history characteristics to refine and retain only the most appropriate group of species among those identified. IndVal analysis identified 17 bird species (six for the protected areas and 11 for the unprotected areas) as potential indicator species for the two land use types across both seasons. Of these, based on the other selection criteria used, four and nine species were finally selected as reliable, appropriate indicator species for the protected and the unprotected areas, respectively. The four species selected for the protected areas were: Brown Woodland Warbler, White-backed Black Tit, Montane White-eye and African Dusky Flycatcher (the former three species being selected in both seasons and the latter only in the dry season), In addition, four species and five species of detectors were identified for the protected areas during the dry and wet season, respectively. Some species were consistently identified as bioindicators across seasons in both habitats, while others were restricted to one season. Identifying different sets of bioindicator species for different seasons helps inform managers as to which species should be* *monitored within a given season when monitoring is carried out. It also helps to avoid poor monitoring outcomes and related management decisions if bioindicator species change from season to season, as demonstrated in this study. The selection of indicator species using the IndVal procedure followed by using other species-specific traits as selection criteria is important to avoid the inclusion of inappropriate species in the indicator species group. Thus, the selected indicators in this way can not only serve for the monitoring of ecosystem health of the Afromontane forests in the Bale Mountains, but also for other similar forests elsewhere, especially in the Ethiopian highlands, that have limited human and financial resources to prioritize monitoring works. Therefore, it is recommended that specifically in protected areas where birds are key ecosystem components and draw tourism, and where funds are limited, a preliminary study on identifying indicator species will maximize conservation efficiency in the long term.*

Keywords: Afromontane forest, Bale Mountains, bioindicators, birds, detector species, disturbance, indicator species, indicator selection criteria.

3.1 Introduction

The creation of protected areas such as national parks, reserves and sanctuaries are critical conservation measures for protecting global biodiversity (Wynne 1998; Locke & Dearden 2005; Jackson et al. 2009). This is because many species are facing extinction risk largely due to habitat loss (Pimm et al. 1995; Brooks et al. 2006). Despite this, many of these protected areas are in danger of losing their biodiversity and ecological value because of increased anthropogenic threats, poor management systems and limited finances (van Schaik et al. 1997;

Struhsaker et al. 2005; Bleher et al. 2006). This is particularly true in developing countries, where conserving biodiversity is often very low on political agendas, and thus receives poor financial support. This is further compounded by exponential human population growth rates (Bruner et al. 2001; Struhsaker 2005). A survey conducted in 201 parks across 16 tropical countries revealed that more than 70% of them are adversely affected by poaching, clearing, encroachment and/or logging (van Schaik et al. 1997). However, generally the level of threats are still lower than in surrounding areas (Bruner et al. 2001) and parks therefore still play a vital role in conserving biodiversity.

Given that a major threat to biodiversity globally is that of habitat loss (Chown 2010), establishing ecological monitoring programmes in protected areas has paramount benefits in implementing effective conservation actions. However, conservation efforts in such protected areas, particularly in developing countries, are limited by constrains typically related to time, funds and expertise, contributing to the impossible task of surveying the distributions of all organisms (Manne & Williams 2003). Therefore, identifying early warning systems for habitat change is a critical measure in helping to mitigate the effects of these threats. An increasingly effective and popular method used to act as an early alarm system is the identification of bioindicators (Noss 1990; McGeoch 1998; Carignan & Villard 2002; Niemi & McDona 2004; Kati & Şekercioğlu 2006). Bioindicators can be defined as a species or group of species that readily reflects the abiotic or biotic state of an environment (i.e. environmental indicators), represents the impact of environmental change on a habitat, community, or ecosystem (i.e. ecological indicators) or is indicative of the diversity of a subset of taxa, or of the wholesale diversity, within an area (i.e. biodiversity indicators; McGeoch 1998; Niemi & McDonald 2004). Previous studies have used bioindicator taxa to assess the impact of human-associated activities on biodiversity (Bock & Webb 1984; Croonquist & Brooks 1989; Temple & Wines 1989), ecosystem health (Bortone & Davis 1994; van Rensburg et al. 1999; Hilty & Merenlender 2000; van Rensburg et al. 2000) and levels of biodiversity and endemism (Faith & Walker 1996; Nally & Fleishman 2002, 2004; Leal et al. 2010).

Two types of bioindicators commonly used for environmental and ecological monitoring purposes are indicator and detector species (McGeoch & Chown 1998; van Rensburg et al. 1999; McGeoch et al. 2002). Indicator species are those 'characteristic' of a particular habitat (i.e. with high specificity and fidelity to a particular habitat and thus a high percentage bioindication value (Dufrene & Legendre1997; McGeoch & Chown, 1998; McGeoch et al. 2002). These species are highly specific (i.e. restricted to a single ecological state) and thus changes in their abundance are useful for monitoring the habitat to which they are specific (Dufrene & Legendre 1997; McGeoch & Chown 1998; van Rensburg et al. 1999). Detector species are those occurring in the different habitats considered but with moderate indication value (McGeoch et al. 2002). Thus when monitoring environmental change, species that span a range of ecological states (i.e. detector species that do not have high specificity), may be more useful indicators of direction of change than highly specific (characteristic) species restricted to a single state (van Rensburg et al. 1999, 2000; McGeoch et al. 2002). Relative changes in abundance of detectors across states may be indicative of the direction in which change is occurring as these species have different degrees of preference for different ecological states (McGeoch & Chown 1998; van Rensburg et al. 1999; McGeoch et al. 2002). Therefore, in order to apply the concept of bioindicators to environmental and ecological monitoring programmes, identification of both indicator species (characteristic species) and detector species (species with some degree of habitat preference) is of paramount importance for monitoring ecological components within a habitat (i.e. to understand the status of a given habitat) and across habitats (i.e. to detect the direction in which change is occurring, McGeoch et al. 2002). Therefore, while indicator species are useful in determining what changes have already taken place within a habitat, from a conservation perspective, detector species are more useful as they can help act as an early warning system.

Birds, among vertebrate groups of animals, have been a primary focus for most terrestrial applications of the bioindicator concept. The primary reasons for their use are: (a) relative ease of identification, (b) relative ease of measurement, (c) relatively large number of species with known responses to disturbance and (d) relatively low cost for monitoring (Niemi & MacDonald 2004).

In this study, I examine the usefulness and applicability of birds as bioindicators in the dry Afromontane forests of the northern Bale Mountains, southeast Ethiopia, and their potential application for forest monitoring and conservation. Deforestation rates, livestock grazing, settlement and agricultural expansion have all increased significantly over recent years in these areas (FZS/BMNP and FA/SOS *unpublished data*; see also Table 1 of Chapter 1). Thus, identifying species that can assist in the long term monitoring of forest status will help by acting as an early warning system and will be an important contribution to the conservation of these areas.

The objectives of this study are therefore to (1) identify bird species that can act as indicator/characteristic species for each land use type (protected and unprotected; and (2) identify bird detector species to indicate and monitor habitat change.

3.2 Materials and Methods

3.2.1 Study area

The Bale Mountains are located in southeast Ethiopia and belong to the Bale-Arsi massif, which forms part of the Ethiopian highlands. At the centre of these mountains is the Bale Mountains National Park (BMNP; 717163- 789904 N, 552126- 604402 E; Fig. 1). The park is 2200 km^2 and encompasses the largest expanse of Afro-alpine habitat on the continent and Ethiopia's second largest tropical montane forest. It ranges in altitude from 1500 to 4377 m a.s.l. (OARDB 2007). The area experiences two rainy seasons, the heavy rain season from July to October and the short rains from March to June. It also comprises a dry season from November – February. Mean annual rainfall is 1219 mm. The area experiences temperature extremities during the dry season and has mean annual minimum and maximum temperatures of 2.36° C and 15.5° C, respectively (OARDB 2007). Five broad vegetation zones occur in the BMNP and the surrounding areas, namely the northern grasslands (a flat area at altitude of 3000 m a.s.l.), the northern dry Afromontane forest (2900 - 3400 m a.s.l.), ericaceous forest (3400 - 3800 m a.s.l.), the Afro-alpine moorland and grassland (3800 - 4377 m a.s.l.), and the southern Harenna forest (1500 - 3200 m a.s.l.; NH 2004; OARDB 2007).

Due to its isolation from the rest of the Ethiopian highlands (Yalden 1983) and the complexity of its habitat, distinct animal and plant species have evolved in the Bale Mountains and there are subsequently very high levels of species richness and endemism (Williams et al. 2004; Asefa 2011; Largen & Spawls 2011). Despite its immense importance as a centre of endemism and evolutionary processes, this region is currently under conservation threat (Stephens et al. 2001; OARDB 2007; Abera & Kinahan 2011; Assefa 2011; Teshome et al. 2011).

The northern part of the BMNP and its surrounding areas comprises patches of dry afromontane forest, characterised by *Juniperus procera*, *Hagenia abysinica* and *Hypericum revoluteum* tree species. Six forest patches of varying size (120 - 2485 ha) and threat level (see Table 1 in Chapter 1), but with similar altitude (2900–3400 m a.s.l.), dominant vegetation and topography were selected for this study. Three patches (Adelay, Boditti and Dinsho Hill) lie inside the protected national park and three outside (Angesso, Shaya and Darkina) in unprotected adjacent areas. Crop production and settlement occurs in the unprotected areas, but not the protected ones (see Table 1 of Chpater 1). Of the 15 transects sampled in the unprotected areas during the present study, houses were encountered along seven of them (47%) and crop cultivation along nine (60%), with mean values of 2.7 houses and 40% cover.ha^{-1}, respectively. Tree logging was significantly higher in the unprotected area than the protected area (number of tree stumps.ha^{-1}: protected area median = 14.06, unprotected area median = 84.38. Similarly, grazing was also found to be significantly higher in the unprotected areas than the protected areas (see Table 1 in Chapter 1, and Chapter 2).

3.2.2 Bird surveys

In each of the six forest patches, five line transects of 1–1.5 km in length were randomly placed parallel to each other (within a range of 200 m – 350 m) along on an altitudinal gradient of 2900 – 3400 m a.s.l., and spaced a minimum of 300 m from each other to reduce pseudo-replication. Along each transect, four fixed bird survey points were selected and point counts conducted (Gibbons et al. 1996; Bibby et al. 1998; Gregory et al. 2004).

Bird surveys were carried out in the dry (November to March) and wet (July to October) seasons of 2009 and 2012, respectively. Each transect in all patches was walked twice in a given day (early morning and late afternoon), thus, each transect was visited eight times in the course of the study. Eight minutes counting time was used with two additional minutes allocated for birds to settle before commencing counting at each point. Within the 8 minutes, birds seen and/or heard within a radius of 50 m were recorded along with their number. Birds flushed away from the census point while approaching the station and those that flew away while counting were recorded from the point they were first seen (van Rensburg et al. 2000; Gregory et al. 2004). Birds that were observed flying over the census area and not necessarily making use of the habitat (e.g. swifts, swallows, scavengers and some raptors) were not recorded. However, some predator birds that hunt from tree canopies (e.g. hawks) and seen making use of the habitat at the time of observation were recorded**.** Counts was conducted early in the morning, between 07h30 and 10h30, and late in the afternoon, between 14h30 and 17h30 when the majority of birds most active.

3.3 Data analysis

3.3.1 Identifying bioindicator species

Indicator species (characteristic bird species) were identified for each land use type using the Indicator Value (IndVal) Method and computed using the IndVal software (Dufrene & Legendre 1997). This assesses the degree (expressed as a percentage) to which each species fulfils the criteria of specificity (uniqueness to a particular site) and fidelity (frequency within that habitat type) for each habitat cluster compared with all other habitats. The higher the percentage IndVal obtained, the higher the specificity and fidelity values for that species, and the more representative the species is of that particular habitat. Species with high IndVal values thus make reliable indicator species not only because they are specific to a locality, but also because they have a high probability of being sampled in that locality during monitoring and assessment (McGeoch & Chown 1998). The species abundance matrix from each site for each season was used to identify indicator species for the protected and unprotected areas for the dry and wet seasons, respectively. The following two comparisons were made: protected area dry season vs. unprotected area dry season and protected area wet season vs. unprotected area wet season. Dufrene & Legendre's (1997) random reallocation procedure of sites among site groups was used to test the significance of the IndVal measures for each species. Those species with statistically significant IndVals > 60% (subjective benchmark) were then regarded as potential indicator species for the habitat in question in each season.

The fact that a given taxon has an IndVal greater than the bench mark does not necessarily mean that it should be used as a reliable indicator of ecosystem health in a monitoring programme because the likelihood of subsequently sampling a given species in the future depends on certain species-specific traits (for example, changes in a population of migratory birds could be due to impacts in any part of their migratory route, not just at the study site) (Hilty & Merelender 2000). Accordingly, appropriate species should be selected using other supplementary criteria in addition to the IndVal approach. These criteria are broadly grouped into four general categories: baseline information, location information, niche and life history characteristics and other (see Hilty & Merelender 2000 for a detailed description of attributes for each criterion; see Table 2 for the criteria and attributes used in this study). Information on these criteria for each species indentified to be potential indicators from the IndVal analysis was obtained from literature surveys (Urban & Brown 1971; Sinclair & Ryan 2003; Redman et al. 2009) and used for refining the final selection of appropriate species.

The protected forest habitats are thought to represent the natural landscape of the northern dry Afromontane forest of the Bale Mountains (OARDB 2007). Disturbance to these patches, such as clearance for crop cultivation, selective logging, settlement establishment and livestock grazing, usually results in an opening up of the habitat and conversion to a mosaic of open land, woodland and dense forest habitat types. To date, no reversion to the original habitat structure has been recorded for disturbed forest patches in the unprotected area (see OARDB 2007). Therefore, in addition to selecting indicator species from IndVal scores, as described above, IndVal scores were also used to identify ‘detector’ species (see McGeoch et al. 2002) in order to detect future changes in the protected area. These were chosen as those that had significant IndVals of 50 - 60% for unprotected forest patches and 5 - 50% for protected forest patches. These species were therefore not characteristic species, as they did not have high IndVals of > 60% for any particular habitat. However, species meeting these criteria were regarded as sufficiently indicative of unprotected forests, but were uncharacteristic of the protected forests at present. Thus, these species were supposed to show a potentially considerable increase in abundance (hence, in indicator value) in the protected forest patches under conditions of increasing disturbance (McGeoch et al. 2002).

The importance of selecting potential detector species using this criterion (i.e. those species with significant IndVals of 50 - 60% for unprotected forest patches and 5 - 50% for protected forest patches) could be justified in different ways. In the first instance, indicator species (i.e. those with high indicator values, and as used here with significant IndVals of >60%) are less likely to move from their preferred habitat to adjacent habitats, even when changes in habitat conditions are present. Populations of these species need only be monitored within their preferred habitat to which they are indicators for. Similarly, owing to their adaptation to a wide rane of habitats, generalist species are also unlikely to respond very rapidly to changing habitat conditions. However, those species with some degree of habitat preference (as used here those with significant IndVal measures of between 50% and 60% from the unprotected forests and less for the protected forests) are likely to move to adjacent habitats more rapidly in response to changes in habitat conditions from one state to other state (e.g. from forest to open woodland) than either indicator or generalist species. These detector species would thus be most likely to invade the protected forest patches in the early stages of their change to a mosaic of open land, woodland and remnant dense forest patches, which is a characteristic of the unprotected forest patches (McGeoch & Chown 1998; van Rensburg et al. 1999; McGeoch et al. 2002). Selection of detector species was carried out separately for the dry season and wet season.

3.4 Results

Of the total 66 bird species recorded in both land use types during both seasons, 60 (52 during the dry season and 53 during the wet season) and 46 (41 during the dry season and 39 during the wet season) were recorded in the unprotected and protected areas, respectively (Appendix I of Chapter 2).

Indval analysis identified 17 bird species (six for the protected areas and 11 for the unprotected areas) as potential indicator species for the two land use types across both seasons. Considering the two seasons separately, fifteen species were identified as indicator species during the dry season and eleven for the wet season, with the same trend observed when examined within a given land use type for each season respectively (protected area: dry season five species, wet season four species; unprotected area: dry season 10 species, wet season 7 species; Table 1). Of the six species identified as indicators of the protected areas, three (50%) of them were identified in both seasons. Similarly, six species (55%) were considered indicator species of the unprotected areas in both seasons (Table 1).

Of the potential indicators identified for the protected areas, two species (Common Chiffchaff (dry season indicator), and Wattled Ibis (wet season indicator) failed to fulfil the criteria to be appropriate indicator species; Common Chiffchaff is a migrant species while Wattled Ibis is a habitat generalist. Similarly, two species (Black-winged Lovebird and Dusky Turtle Dove) did not fulfil the criteria of habitat specificity. Thus, four species were selected as reliable indicator species for the protected areas and nine species for the unprotected areas (Table 2).

Table 1 Species identified as indicators for the protected and unprotected areas for the wet and dry seasons with their % IndVal values (all IndVal values were significant at the alpha 0.05 level). Species in bold were omitted during the final selection.

Site	Dry season		Wet season	
	Species	IndVal	Species	IndVal
Protected area	Brown Woodland Warbler	61.04	Montane White-eye	61.30
	White-backed Black Tit	76.99	**Wattled Ibis**	61.30
	Montane White-eye	73.96	Brown Woodland Warbler	66.54
	African Dusky Flycatcher	62.88	White-backed Black Tit	78.75
	Common Chiffchaff	91.94		
Unprotected area	White-cheeked Turaco	60.32	**Dusky Turtle Dove**	64.47
	Black-winged Lovebird	60.70	Tawny-flanked Prinia	67.31
	Brown-rumped Seedeater	62.86	Olive Thrush	67.76
	Ground Scraper Thrush	63.25	Brown-rumped Seedeater	70.14
	Cinnamon-bracken warbler	64.41	Cinnamon-bracken Warbler	72.05
	Olive Thrush	65.67	Streaky Seedeater	73.50
	Moorland Chat	67.80	Common Bulbul	85.25
	Tawny-flanked Prinia	68.09		
	Common Bulbul	68.87		
	Streaky Seedeater	73.30		

Table 2. Criteria and attributes used for the final selection of appropriate indicator species among those identified using the IndVal procedure. Numerical codes under each species column shows the applicability of each attribute for that species (1 = applicable; 0 = not applicable)

		species								
Criteria	attributes	Brown Woodland Warbler	African Dusky Flycatcher	Montane White-eye	White-backed Black Tit	Common Chiffchaff	Wattled Ibis	White-cheeked Turaco	Black-winged Lovebird	Brown-rumped Seedeater
Baseline information	species with clear taxonomic status	1	1	1	1	1	1	1	1	1
Location information	species with wide distribution, (i.e. national, regional or global distribution)	1	1	1	1	1	1	1	1	1
	non-migrant	1	1	1	1	0	1	1	1	1
Niche and life histories	habitat specialist	1	0	1	1	1	0	1	0	1
	easy to find and measure/high abundance	1	1	1	1	1	1	1	1	1

Table 2. Continued…

		species							
Criteria	attributes	Ground Scraper Thrush	Cinnamon-bracken warbler	Olive Thrush	Moorland Chat	Tawny-flanked Prinia	Common Bulbul	Streaky Seedeater	Dusky Turtle Dove
Baseline information	species with clear taxonomic status;	1	1	1	1	1	1	1	1
Location information	species with wide distribution, regional or global distribution	1	1	1	1	1	1	1	1
	non-migrant	1	1	1	1	1	1	1	1
Niche and history	habitat specialist	1	1	1	1	1	1	1	0
	easy to find and measure/high abundance	1	1	1	1	1	1	1	1

Nine species (four during the dry season and five during the wet season) were identified as detectors for the protected areas (Table 3). Unlike, indicator species, none were identified as detectors in both seasons.

Table 3. Species identified as detectors for the protected areas during the dry and wet seasons and their %IndVal values (all IndVal values are significant at the P ≤ 0.05 level).

Dry season		Wet season	
Species	%IndVal values	Species	%IndVal values
Abyssinian Ground Thrush	50.2	Baglafecht Weaver	53.33
Tropical Boubou	52.78	Abyssinian Woodpecker	46.45
Tacazze Sunbird	56.04	Slender-billed Starling	58.29
Cape Crow	53.33	Moorland Chat	57.14
		Red-throated Wryneck	40.44

3.5 Discussion

The two land types (protected and unprotected forest patches) differed in disturbance levels, and consequently bird assemblages, as shown in Chapter 2. Thus, given these avian assemblage differences resulting from these varying levels of disturbance, there is a high likelihood that both indicator and detector species can be identified and used to assist with the monitoring and conservation of the dry Afromontane forest as well as the unique bird diversity.

The unprotected areas had higher numbers of characteristic species than the protected areas, possibly due to the dominance of its assemblage by openland and shrubland habitat species, several of which are specific to that habitat. In contrast, although the protected area assemblage is dominated by forest-specialist species (Chapter 2), a large number of these species also occur within the unprotected forest areas. Thus, more species occurred sufficiently frequently or abundantly to be identified as characteristic indicators of unprotected compared to protected areas. Some species were consistently identified as indicators across seasons in both habitats, while others were restricted to one season. None of the detectors identified for the protected areas were selected in both seasons. This could indicate differential responses of species in assemblages where populations of some species show a seasonal shift in habitat use as a consequence of changes in habitat conditions. This is especially true of the unprotected areas where seasonal change in land use occurs each year. During the wet season, most of the area is covered by crops, attracting shrubland species but negatively affecting openland species and *vice versa* during dry season. Moreover, the identification of different sets of bioindicator species for different seasons has significant importance from a monitoring point of view. For example, a budget may only allow for monitoring once a year, thus within a given season, and for this it will be important to know which species should be monitored within the given season when the monitoring is conducted. Further, if bioindicator species were to change from season to season, as is the case demonstrated in this study, it could lead to poor monitoring outcomes, and related management decisions, if the set of bioindicator species used are not relevant (or accurate) for the time of year when the monitoring is conducted.

Four species, namely Brown Woodland Warbler, White-backed Black Tit, Montane White-eye and African Dusky Flycatcher (the former three species being selected in both seasons and the latter only in the dry season), were selected as appropriate indicator species for the protected area based on their indicator values and merits of fulfilling the supplementary criteria used. All of these species have large distribution ranges, except the White-backed Black Tit which is found only in the highlands of Ethiopia and Eritrea (Sinclair & Ryan 2003; Redman et al. 2009), and are considered as least concern in terms of conservation status within their ranges

(BirdLife International 2013). White-backed Black Tit and Montane White-eye are the only species of their genus so far reported from the northern dry Afromontane forest of the Bale Mountains (Hillman, 1986; Asefa, 2006/07), and Brown Woodland Warbler and African Dusky Flycatcher can easily be distinguished from their co-occurring congeneric species (i.e. Common Chiffchaff and Abyssinian Slaty Flycatcher, respectively; see Sinclair & Ryan 2003; Redman et al. 2009), making their identification by both non- bird specialists and specialists easy. It is these four species, therefore, that would be the most useful for monitoring the protected areas over the long term.

Despite the wide application of indicator taxa to monitor ecosystem health, ambiguous selection criteria and the use of inappropriate taxa have brought the utility of indicator taxa into question (Hilty & Merenlender 2000). Consequently, using a combination of different approaches, which are supplementary to each other and not mutually exclusive, as used in the present study, could help overcome the pitfalls of selecting inappropriate indicator species. The rationale for using additional criteria to refine the selection of indicator species is that species proposed to be potential indicators by virtue of their higher IndVal values alone may lead to the inclusion of inappropriate species in indicator taxa groups. For instance, location information is important in selecting indicator taxa; selected taxa should have a cosmopolitan (national, continental or global) distribution to assist in cross-comparisons of sites (Hellawell 1986; Noss 1990; Regier 1990; Pearson & Cassola 1992; Johnson et al. 1993). In the present study, the selected indicator species can be used, at least, in the Ethiopian highlands and potentially in other areas of their range. An indicator taxon should also have limited mobility, so that it is less likely to be able to avoid disturbances (Landres et al. 1988; Johnson et al. 1993). For example, changes in a population of migratory birds could be due to impacts in any part of their migratory route, not just at the study site. Specified niche history characteristics, like habitat specialization, are considered an important criterion because generalists can potentially avoid impacts by altering their habitat use, thereby failing to respond to the impact. A set of complementary specialist indicator taxa representative of a range of ecosystem niches would potentially allow for detection of fine-scale impacts and early detection, while use of multiple generalists is unlikely to lead to such enhanced fine-scale early detection.

The detector species that were identified for the protected areas are ideal species to monitor the direction of habitat change impacts in the protected areas. These species tend to be uncommon in the protected area (and generally widespread in the unprotected area) at present. However, if the protected forest were be impacted, it is assumed that the indicator species will decline or locally disappear while the detectors colonize the site. Therefore, detecting the presence of a new species (i.e. a detector species) and an increase in its frequency and abundance in protected areas in the future is likely to be far more reliably undertaken than detecting the absence of a species (i.e. indicator species) (McGeoch 1998; van Rensburg et al. 1999; McGeoch et al. 2002). Thus, the most useful species for indicating change in the protected area habitats are both those with a high IndVal for the protected areas (i.e. indicator species), and those species currently present in unprotected areas and identified as detector species for the protected areas (van Rensburg et al. 1999; McGeoch et al. 2002). Similar to the indicator species, all the detectors have wide distributions (with the exception of Abyssinian Woodpecker, a species endemic to Ethiopian montane forest (Redman et al. 2009), and are categorized as least concern conservation status (BirdLife International 2013). Therefore, they can potentially be applied for monitoring in the study area, as well elsewhere in their range.

The increasing need to make appropriate conservation decisions based on scientific evidence, as well as the presence of resource constraints limiting data collection, necessitates protected area managers to develop simple, repeatable and cost-effective approaches for assessment of ecosystem health. The selection of reliable bioindicators is one such approach being implemented in ecological monitoring purposes in several protected areas globally (Lindenmaye 1999; Hilty & Merenlender 2000). Thus, the species identified as indicators for the protected area in the present study can provide useful information for forest managers as an early warning of change. Their use for ecological monitoring can be enhanced by using detector species as the latter provide supplementary information about the state of the system concerned. In order to practically use the proposed indicators and detectors in the present study for monitoring of habitat condition of the protected forest patches, both categories of species should be counted following same procedures used in this study. Counting could be done every two to three years (depending on the resources availability) in each season, and changes in their abundance and/density could be compared over years. This is particularly interesting as the BMNP, with a technical assistance from FZS-BMCP, has recently launched ‘Ecological and Threat Monitoring Programme’ in some prioritized ecosystems of the park, including the present study area (BMNP/FZS BMCP, unpublished documents). Therefore, information obtained from monitoring of the bioindicators can be related to the various threats to the forests, so that they would be reliably applied and used for management decisions. However, prior to using the bioindicators identified here, testing (verification of) the validity of them is required by testing the robustness of identifying similar species over a temporal scale (McGeoch et al. 2002).

3.6 Conclusion

Using the IndVal procedure we were able to identify indicator species and detector species for habitat change in protected Afromontane forests of the Bale Mountains. The applicability of this tool is not only useful for the Bale Mountains National Park, but also for other protected areas in developing countries that have limited human and financial resources to prioritize monitoring. The information attained through this method is directly applicable and useful for management decisions. Therefore, it is recommended that in protected areas specifically where birds are a key tourism draw card, and where funds are limited, a preliminary study on identifying indicator species would maximize conservation efficiency in the long run. More specifically, given the lack of forest monitoring programmes in the study area, but ever-increasing forest conversion (see Abera & Kinahan 2011 for annual rate of forest loss in the Bale Mountains), the present findings can be incorporated into the on-going ecological monitoring programme in the BMNP (see OARDB 2007, for detailed information about the monitoring programmes running in the national park). However, prior to developing an ecological monitoring protocol using the bioindicators identified here, testing (verification of) the validity of them is required by testing the robustness of identifying similar species over a temporal scale. This validation can be done either by sampling from the same points from where the present data were collected or by sampling from other sites using the same technique (McGeoch et al. 2002).

3.7 References

Abera, K., and A.A. Kinahan. 2011. Factors Affecting Fire Extent and Frequency in the Bale Mountains National Park. Walia-Special Edition on the Bale Mountains 146-157.

Asefa, A. 2011. Mammals of the Bale Mountains National Park, Ethiopia: a compiled and annotated checklist. Walia-Special Edition on the Bale Mountains 3-14.

Bibby, C., M. Jones, and S. Marsden. 1998. Expedition field techniques: bird surveys. The expedition Advisory Centre, Royal Geographical Society. The institute of British geographers, London.

BirdLife International. 2013. Country profile: Ethiopia. Available from: http://www.birdlife.org/datazone/country/ethipia. checked: 07/05/2013

Bleher, B., D. Uster, and T. Bergsdorf. 2006. Assessment of threat status and management effectiveness in Kakamega Forest, Kenya. Biodiversity and Conservation **15**:1159–1177.

Bock, C.E., and B. Webb. 1984. Birds as grazing indicator species in the south-eastern Arizona. Journal of Wildlife Management **48**:1045-1049

Bortone, S.A., and W.P. Davis. 1994. Fish intersexuality as indicator of environmental stress. Bioscience **44**:165-172.

Brooks, T.M., R. A. Mittermeier, G. A. B. da Fonseca, J. Gerlach, M. Hoffmann, J. F. Lamoreux, C. G. Mittermeier, J. D. Pilgrim, and A. S. L. Rodrigues. 2006. Global biodiversity conservation priorities. Science **313**: 58-63.

Bruner, A.G., R.E. Gullison, R.E. Rice, and G.A.B. da Fonseca. 2001. Effectiveness of Parks in Protecting Tropical Biodiversity. Science **291**: 125-127

Carignan, V., and M. Villard. 2002. Selecting indicator species to monitor ecological integrity: a review. Environmental Monitoring and Assessment. **78**: 45–61.

Chown, S.L. 2010. Temporal biodiversity change in transformed landscapes: a southern African perspective. Philosophical Transactions of the Royal Society, Biological sciences **365**: 3729-3742.

Croonquist, M.J., andR.P. Brooks. 1991. Use of avian and mammalian guilds as indicators of cumulative impacts in riparian wetland areas. Environmental Management **15**: 701-714.

Dufrene, M., and P. Legendre. 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecological Monographs **67**: 345-366.

Faith, D.P., and P.A. Walker. 1996. Environmental diversity: on the best possible use of surrogate data for assessing the relative biodiversity of sets of areas. Biodiversity and Conservation **5**: 399-415.

Gibbons, D.W., D.A. Hill, and J.W. Sutherland. 1996. Birds, in: Sutherland, J.W. (ed.), Ecological census techniques: A hand book. Cambridge University Press, Cambridge, pp. 227-259.

Gregory, R.d., Gibbons, D.W., and P.F. Donald. 2004. Bird census and survey techniques, in: Sutherland, W.J., I. Newton, and R. Green. (eds.), Bird ecology and conservation: a handbook of techniques –Issue 1 of techniques in ecology and conservation series. Oxford University Press, Oxford, pp. 17-56.

Hilty, J., and A. Merenlender, 2000. Faunal indicator taxa selection for monitoring ecosystem health. Biological Conservation **92**:185-197.

Jackson, S.F., K. Walker, and K.J. Gaston. 2009. Relationship between distributions of threatened plants and protected areas in Britain. Biological Conservation **142**:1515–1522.

Kati, V.I., and C.H. Şekercioğlu. 2006. Diversity, ecological structure, and conservation of the landbird community of Dadia reserve, Greece. Diversity and Distributions**12**: 620–629

Largen, M., and S., Spawls. 2011. Amphibians and Reptiles Recorded from the Bale Mountains. Walia-Special Edition on the Bale Mountains 89–91.

Leal, I. R., A.G. D. Bieber, M. Tabarelli, and A.N. Andersen. 2010. Biodiversity surrogacy: indicator taxa as predictors of total species richness in Brazilian Atlantic forest and Caatinga. Biodiversity Conservation **19**: 3347–3360.

Locke, H., and P. Dearden. 2005. Rethinking protected area categories and the new paradigm. Environmental Conservation **32**: 1–10.

Manne, L.L., and P.H. Williams. 2003. Building indicator groups based on species characteristics can improve conservation planning. Animal Conservation **6**: 291–297.

McGeoch, M. A., and S. L. Chown. 1998. Scaling up the value of bioindicators. Tree **13**: 46-47

McGeoch, M.A., B. J. Van Rensburg, and A., Botes. 2002. The Verification and application of Bioindicators: A Case Study of Dung Beetles in a Savanna Ecosystem. Applied Ecology **39**: 661-672.

McGeoch, M.A. 1998. The selection, testing and application of terrestrial insects as bioindicators. Biological Reviews **73**: 181-201.

Nally, R.M., and E. Fleishman. 2004. A successful predictive model of species richness based on indicator species. Conservation Biology **18**: 146-154

Newton, A.C. 2007. Forest ecology and conservation: a handbook of techniques in ecology and conservation series. Oxford University Press, Oxford.

NH (The National Herbarium), 2004. Biodiversity assessment of the Bale Mountains National Park and surrounding areas. The National Herbarium of Addis Ababa University, Addis Ababa.

Niemi, G.J., and M.E. McDonald. 2004 Application of ecological indicators. Annual Review in Ecology, Evolution and Systematics **35**: 89-111

Noss, R.F. 1990. Indicators for monitoring biodiversity: a hierarchical approach. Conservation Biology **4**: 355-364.

OARDB. 2007. Bale Mountains National Park General Management Plan. Oromia Agriculture and Rural Development Bureau, Addis Ababa.

Pimm, S.L., G.J. Russell, J.L. Gittleman and T.M. Brooks. 1995. The future of biodiversity. Science **269**: 347–350.

Redman, N., T. Stevenson, and J.,Fanshawe. 2009. Birds of the Horn of Africa: Ethiopia, Eritrea, Djibouti, Somalia and Socotra. Christopher Helm, London.

Sinclair, I., and P., Ryan. 2003. Birds of Africa south of the Sahara: A comprehensive illustrated field guide. Struik publishers, Cape Town, South Africa.

Stephens, P.A., C.A. D'sa, C. Sillero-Zubri, and N.L. Williams. 2001. Impact of livestock and settlement on the large mammalian wildlife of Bale Mountains National Park, Southern Ethiopia. Biological Conservation **100**: 307-322.

Struhsaker, T.T., P.J. Struhsaker, and K.S. Siex. 2005. Conserving Africa's rain forests: problems in protected areas and possible solutions. Biological Conservation **123**: 45-54.

Temple, S.A., and J.A. Wiens. 1989. Bird populations and environmental changes: can birds be bio-indicators. American Birds **43**: 260-270.

Teshome, E., D. Randal, and A.A. Kinahan. 2011. The Changing Face of the Bale Mountains National Park over 32 years: A Study of Land Cover Change. Walia-Special Edition on the Bale Mountains 118-130.

Urban, E. K., and L. E. Brown. 1971. A checklist of the birds of Ethiopia. Haile Sellassie I University Press, Addis Ababa, Ethiopia

Van Rensburg, B. J., M. A. McGeoch, S. L. Chown, and A.S. Van Jaarsveld. 1999. Conservation of heterogeneity among dung beetles in the Maputaland Centre of Endemism, South Africa. Biological Conservation **88**:145-153.

van Rensburg, B.J., S.L. Chown, A.S. van Jaarsveld, M.A. McGeoch. 2000. Spatial variation and biogeography of sand forest avian assemblages in South Africa. Journal of Biogeography **27**: 1385–1401

van Schaik C.P., J. Terborgh, and D. Dugelby. 1997. The silent crisis: the state of the rainforest nature reserves. In: Kramer R., C.P. van Schaik, and J. Johnson (eds), The Last Stand: Protected Areas and the Defense of Tropical Biodiversity. Oxford University Press, New York, pp. 64–89.

Williams, S., J.L. Vivero Pol, S. Spawls, A. Shimelis, E. Kelbessa. 2004. Ethiopian Highlands, in: Mittermeier, R.A., Gill, P.R., Hoffmann, M., Pilgrim, J., Brooks, T., Mittermeier, C.G., Lamoreux, J., Da Fonseca, G.A.B. (eds.), Hotspots revisited. CEMEX Publisher, Mexico City, pp. 262-273.

Wynne, G. 1998. Conservation policy and politics, in: Sutherland, W.J. (ed.), Conservation, Science and Action. Blackwell Science, Oxford, pp. 256-285.

Yalden, D.W. 1983. The extent of high ground in Ethiopia compared to the rest of Africa. SINET: Ethiopian Journal of Sciences **6**: 256-285.

Chapter 4: Conclusions and recommendations

This dissertation has contributed to our understanding of the impacts of human disturbance on Afromontane avian assemblages and on selection techniques of appropriate indicator species, particularly birds, for the monitoring of forest ecosystem health. Previous studies (e.g. Recher 1969; Trzcinski et al. 1999; Şekercioğlu 2002; Heikkinen et al. 2004; Kessler et al. 2005; Kumar & Ram 2005; Tabeni and Ojeda 2005; Chown 2010; Mengesha et al. 2011) quantifying the impacts of human disturbances on biodiversity have found complex results, suggesting that impacts can be either positive or negative, depending on the type and severity of the disturbance, type of habitat and the taxa considered. Thus, this dissertation was aimed at assessing the impact of land use change on birds by comparing differences in their diversity (species richness, abundance and assemblages composition) between protected and unprotected Afromontane forest patches of the Bale Mountains, Ethiopia, and to identify bioindicator and detector species for the monitoring of ecosystem health in the protected forests.

The two land use types did not only differ in the type and level of disturbance, but also in habitat variables. Crop production and settlement were recorded only in the unprotected areas, and logging and grazing were also significantly higher in the unprotected areas than the protected areas. Overall bird species richness was significantly higher in the unprotected areas than the protected areas when seasons were treated separately and seasonal data were combined, while no significant difference was detected in terms of abundance. At the guild level, woodland and open land habitat guilds, granivore and insectivore feeding guilds, and shrub layer and ground layer foraging substrate guilds had significantly higher numbers of species in the unprotected areas than the protected areas. Most of these guilds (e.g. woodland and openland habitat guilds, and granivore guilds) had significantly higher abundances in the unprotected areas than the protected areas. In contrast, the canopy layer foraging guild had significantly higher abundance of birds in the protected areas. The differences in species composition of bird communities between the protected and unprotected areas were very distinct, and ecosystem processes such as resource (e.g. food and habitat) availability and inter- and/or intra-specific interactions are likely the underlying factors for the differences found. In general, assemblages in the protected areas are dominated by forest-specialist guilds, whilst the unprotected areas are dominated by openland and shrubland guilds. This indicates that habitat disturbance has caused the invasion of non-native species, and negatively affected forest-specialist species. This study has demonstrated that bird species abundance in the protected and unprotected forests can be predicted by habitat variables within each habitat type, consistent with the habitat heterogeneity hypothesis (Tews et al. 2004).

Bioindicator species were selected using the IndVal procedure (Dufrene & Legendre 1997), and species selection was then refined with other appropriate criteria (see Hilty & Merenlender 2000) to avoid the inclusion of inappropriate species in the indicator species group. The selection of bioindicators in this way provides a useful tool for managers of Afromontane forest in the Bale Mountains, assisting in the long term monitoring of ecosystem health. The selected indicators can also serve for monitoring in other Afromontane forests ecosystems, especially in the Ethiopian highlands.

Conservation implications

One of the main goals of protecting the Afromontane forest of the Bale Mountains is to maintain bird species diversity typical of that habitat (i.e. forest species; see OBARD 2007). However, the present findings that disturbance has caused encroachment of non-native species (openland, woodland and shrub land species), and negatively affected the native avian fauna (forest species, particularly tree canopy foragers) implies that local extinction of forest specialists will occur if effective conservation efforts are not implemented and the rate of forest destruction in the Bale Mountains continues at the current rate. Thus, further forest degradation in the protected forest patches should be avoided in order to maintain native/forest-specialist species. In addition, promoting some sort of conservation initiatives (e.g. community-based conservation projects) in the unprotected areas could help prevent further deterioration of the forests, thereby ensuring the conservation of the full-array of species in this Afromontane landscape.

Given the financial constraints and lack of expertise (OBARD 2007) faced by the managers of the Bale Mountains, it could be impractical to monitor all the ecosystem components of the forests. Therefore, the selected bioindicator and detector species provide a useful tool for managers of the forests for long term monitoring of ecosystem health. Prior to applying this bioindicator concept for the monitoring of ecosystem health of the forests, however, additional data representing a temporal scale should be collected from the area, or from other similar areas, using the same methods in order to verify the usefulness of the selected species (McGeoch & Chown 1998, McGeoch et al. 2002) and determine sample sizes needed for the monitoring and frequency of data collection.

Suggested future research

The impact of human disturbance on the conservation of biodiversity has not been adequately addressed. In order to further enhance our understanding of the impact of disturbance on bird assemblages and of using birds as bioindicators for ecological monitoring, the following future research directions are suggested:

- More comparable studies focusing on disturbance-bird relationships across different habitats. This should involve sites with varying levels of disturbance across different regions.
- Most previous studies conducted to examine the impact of disturbance on birds have compared protected/intact habitats with the surrounding unprotected/disturbed habitats. However, apart from disturbance, there are a number of factors (e.g. the type of surrounding matrix, patch size and patch isolation) that determine the presence of a species that is a member of a regional species pool in a given habitat. Therefore, comparable studies focusing on areas where unprotected sites are surrounded by protected areas would be useful to address some aspects of the current controversy surrounding disturbance-bird relationships.
- While comparing different habitats in terms of bird guild composition, prior subjective assignment of species guild membership based on broad, qualitative information derived from secondary sources could lead to the placement of species in the incorrect guilds, potentially resulting in erroneous conclusions drawn from comparisons involving such guilds. Therefore, posterior guild classification based on quantitative data collected on relevant behavioural activities of individuals of each species (e.g. habitat and substrate use, food type consumed) in the relevant areas is important to accurately determine species guild membership. This does not only help make valid comparisons of distribution and

abundances of species/guilds among different habitats, but also helps explain relationships in the patterns observed with the underlying processes.

- Reviews of previous studies (e.g. Hilty & Merenlender 2000) have indicated that there is little overlap in the criteria that has been used for selecting bioindicator taxa. Although the IndVal procedure is advocated to be one of the most suitable techniques for identifying bioindicators (Dufrene & Legendre 1997; McGeoch et al. 2002), Hilty & Merenlender (2000) suggested the use of other criteria, which are supplementary to the IndVal procedure, to select reliable indicator taxa. Subsequently, studies comparing the performance of bioindicators selected using the IndVal procedure alone and when a combination of IndVal and other criteria are used (Chapter 3) will make a useful contribution to our scientific understanding of bioindicator species selection.

References

Chown, S.L. 2010. Temporal biodiversity change in transformed landscapes: a southern African perspective. Philosophical Transactions of the Royal Society, Biological sciences **365**: 3729-3742.

Dufrene, M. and P. Legendre. 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecological Monographs **67**: 345-366.

Heikkine, R., M. Luoto, R. Virkkala, and K. Rainio. 2004. Effects of habitat cover, landscape structure and spatial variables on the abundance of birds in agricultural-forest mosaic. Journal of Applied Ecology **41**: 824-835.

Hilty, J., and A. Merenlender. 2000. Faunal indicator taxa selection for monitoring ecosystem health. Biological Conservation **92**:185-197.

Kessler, M., P.J.A. Kebler, S.R. Grandstein, K. Bach, M. Schmill, and R. Pitopang. 2005. Tree diversity in primary forest and different land use systems in central Sulawes, Indonesia. Biodiversity and Conservation **14**: 547-560.

Kumar, A., and J. Ram. 2005. Anthropogenic disturbances and plant biodiversity in forests of Uttaranchal, Central Himalaya. Biodiversity and Conservation **14**: 309-331.

McGeoch, M. A., and S. L. Chown. 1998. Scaling up the value of bioindicators. Trends in Ecology and Evolution **13**: 46-47

McGeoch, M.A., B. J. Van Rensburg, and A. Botes. 2002. The Verification and application of Bioindicators: A Case Study of Dung Beetles in a Savanna Ecosystem. Applied Ecology **39**: 661-672.

Mengesha, G., Y. Mamo, and A. Bekele. 2011. A comparison of terrestrial bird community structure in the undisturbed and disturbed areas of the Abijata Shalla lakes national park, Ethiopia. International Journal of Biodiversity and Conservation **3**: 389-404.

OARDB. 2007. Bale Mountains National Park General Management Plan. Oromia Agriculture and Rural Development Bureau, Finfinnee, Ethiopia.

Recher, H.F. 1969. Bird species diversity in Australia and North America. The American Naturalist **103**: 75-80.

Şekercioğlu, C.H. 2002. Effects of forestry practices on vegetation structure and bird community of Kibale National Park, Uganda. Biological Conservation **107**: 229-240

Tabeni, S. and R.A. Ojeda. 2005. Ecology of the Monte Desert small mammals in disturbed and undisturbed habitats. Journal of Arid Environments **63**: 244-255

Tews, J., U. Brose, V. Grimm, K. Tielbörger, M. C. Wichmann, M. Schwager, and F. Jeltsch. 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. Journal of Biogeography **31**: 79-92.

Trscinski, M.K., L. Fahrig, and G. Merriam. 1999. Independent effects of forest cover and fragmentation on the distribution of forest breeding birds. Ecological applications **9**: 586-593.