University of Nairobi

Monitoring Habitat at Key Biodiversity Sites in Africa using Remote Sensing:

Land Cover Change at Important Bird Areas in Eastern Africa

By

Eshiamwata, George Were

B.Sc (Hons) Natural Resource Management (Egerton), M.Sc. Conservation Biology (Cape Town)

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School of Engineering

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Declaration

I hereby solemnly declare that the work presented in this dissertation is my original work and that, to the best of my knowledge, it has not been presented for a degree in any other University

Signed.............................. **Date.** 11·04·2012
George Were Eshiamwata
(Candidate)

This Thesis has been submitted for examination with our approval as University Supervisors

Signed.............................. **Date.** 14-5-2012
Prof. Galcano C. Mulaku
(Lead Supervisor)

Signed.............................. **Date.** 09. 05. 2012
Prof. John B.K. Kiema
(Supervisor)

Signed.............................. **Date.** 4/4/2012
Dr. Graeme Buchanan
(Supervisor)

Dedication

To my wife and best friend Edith, daughters Karen, Trixie and sons Allan and Jasper for their patience and unwavering support throughout this studentship. It is also posthumously dedicated to my late parents, (Emily and Julius). This thesis is in memory of my father who, after encouraging and inspiring me to work hard in life, died just a month before I submitted it. Special dedication is also extended to my aunt Winfred, who is my heroine as she ably filled in my heart the position and void left by my mother.

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Abstract

Land cover change related to both anthropogenic activity and natural causes is taking place invariably at different time periods, paces and magnitudes and with diverse biophysical implications on the various land cover types as well as on biodiversity. This study investigated the percentage and extent of land cover change across all major land cover types at a subset of 71 Important Bird Areas in Eastern Africa over a 20-year period using remotely sensed data. At regional level, during the study period, broad land cover change trends show a dramatic decline in natural habitats such as forests, shrub lands, herbaceous and flooded vegetation. The decline in natural land cover types and a concomitant increase in the percentage of modified and converted land cover types points to the fact that natural habitats were converted or modified into anthropogenic and intensively managed land cover and land uses types such as agriculture, plantation crops, natural agricultural mosaic and urban areas. Population density at the sites varied between 9 and 2,701 persons per sq.km with the latter associated with sites in proximity to urban centres. Population density around IBAs was not a significant correlate of extent and rates of land cover change within IBAs for all land cover types except that it slightly contributed to change in shrub land. Outside IBAs, population density was, as was expected, the most significant factor driving urban expansion, probably manifested by the conversion of natural habitats to urban landscape. Agriculture was the most significant correlate of land cover change driving the observed changes in all forests, open forest/woodlands, shrub lands and herbaceous/grasslands. It was not significant for the rest of the land cover types. Regarding land cover change outside IBAs, agricultural extent was the most significant factor in natural-agricultural mosaic, herbaceous/grassland and urban land cover changes. The impact of legal protection and site specific past conservation interventions was assessed. Protection was not a significant factor in reducing land cover type apart from open forest/woodland habitats. Working with Site Support Groups (SSGs) to implement site specific conservation interventions did not translate into reduced land cover change at IBAs across all land cover types apart from shrub land. However, whether this could be attributed to the smaller sample size of the SSGs used in this study and the shorter period some of the SSGs have been in existence requires further assessment with larger sample sites with a longer history of engagement. All forests and grasslands IBAs were identified as the key biodiversity sites most vulnerable to increased rates of deleterious land cover change as a result of agricultural expansion and intensification and human population density. It is recommended that urgent measures be put in place to improve effective management of protected areas to increase their ability to conserve biodiversity. Efforts should also be expended towards preventing deleterious land cover change outside key biodiversity sites through sustainable land use systems and environmental friendly development initiatives. This will reduce the effects of fragmentation and edge effects and promote the conservation of biodiversity within the surrounding agricultural and human dominated and modified landscapes. This study represents a prototype technique that can be applied to IBAs elsewhere.

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Abbreviations and Acronyms

AEWA	African-Eurasian Waterbirds Agreement
CBD	Convention on Biological Diversity
CMS	Convention on Migratory Species
EBA	Endemic Bird Area
EC-JRC	European Commission- Joint Research Centre
FAO	United Nations Food and Agricultural Organisation
GIS	Geographical Information System
GUI	Graphical User Interface
HPD	Human Population Density
IBA	Important Bird Area
ICPB	International Council for Bird Preservation
IUCN	International Union for Conservation of Nature
KBA	Key Biodiversity Area
MEA	Multilateral Environmental Agreements
METT	Management Effectiveness Tracking Tool
RSPB	Royal Society for the Protection of Birds UK
SSG	Site Support Group
UNCCD	United Nations Convention to Combat Desertification
UNEP	United Nations Environmental Programme
UNESCO	United Nations Educational, Scientific and Cultural Organisation
UNFCCC	United Nations Framework Convention on Climatic Change
UNPD	United Nations Population Division
WCMC	World Conservation Monitoring Centre
WWF	The World Wide Fund for Nature

CHAPTER ONE: INTRODUCTION

1.1. Background to the Study

1.1.1. Land cover change

Extensive land degradation and detrimental habitat changes continue to increase in many parts of the world (Lindenmayer *et al.,* 2002) with increasing documentation of this loss and degradation from a range of habitats (e.g. Goriup, 1988; Moser *et al.,*1996; UNEP-CBD, 2001; Geist and Lambin, 2002; Millington *et al.,* 2003; Wade *et al.,* 2003). The pace, magnitude and spatial breadth at which global biodiversity is changing is unprecedented (Pimm *et. al.,* 1995), the most important drivers of this change being land conversion, climate change, pollution, unsustainable harvesting of natural resources and the introduction of exotic species (Sala *et al.,* 2000).

According to Milne (1988), change is tyhe alteration in the surface components of the vegetation cover. Ecosyems are continously changing and the change can either be dramatic and/or , abrupt (e.g as exemplified by fire) or subtle and/or gradual (Coppin *et al.,* 2004). Anthropogenic land cover changes, whether direct (e.g. deforestation, agricultural expansion) or indirect (e.g. anthropogenic climate change) is the greatest threat to biodiversity (Brooks *et al.,* 2002; BirdLife International 2004; 2008) and these have accelerated natural background extinction rates (Pimm *et al.,* 1995). Human activities are altering the terrestrial environment at unprecedented rates, magnitudes and spatial scales (Turner *et al.,* 1994) such that the current rates, extents and intensities of land cover and land use change are far greater than ever in history, driving unprecedented changes in ecosystems and environmental processes at the local, regional and global scale.

The world is a constantly and unpredictably changing place and land cover change is driven by various multiple factors either operating singly or collectively at different temporal and spatial scales. Various threats operate differently and the impact they have on the rates of land cover change may also vary significantly across regions, countries, habitat types and in time. The threats to biodiversity occur in various forms including direct and indirect underlying factors.

1.1.2. Correlates of land cover change: Human population

Population dynamics have been known to affect biodiversity in powerful and often detrimental ways. The loss in biodiversity has been linked to the exponential rate of growth in human population (Gehrt, 1996; Maurer, 1996; Balmford *et al.*, 2001b; Harcourt and Sparks, 2003). The combination of increasing human numbers and increased consumption, together with human land use patterns, are changing the planet on a scale and with a speed that is unprecedented. Thus population has in turn exerted great pressure on natural resources owing to increased demand for food, land for settlement, water, energy and other resources (Lori, 2000). Studies (Balmford *et al.*, 2001a, b; Luck *et al.*, 2004) have shown the existing congruence between species richness and human population density and that species richness for many taxonomic groups is often highest in areas with high human population density (HPD). This therefore means that if population density related factors are to drive biodiversity loss, the impact would be felt more where species richness is high.

Studies have been conducted to examine the relationships between human population density and biodiversity change (Luck, 2007). Considering that human population is predicted to rise to between 8 and 10 billion by 2050 (Lutz *et al.*, 2001; UNPD, 2003), its impact on land cover change and biodiversity conservation is a major issue that requires constant monitoring.

1.1.3. Correlates of land cover change: Agricultural intensity

It is widely recognised that change in agricultural land use and intensity is an important driver of biodiversity loss in developing countries (Sala *et al.,* 2000) and the general effects of the conversion to agriculture of natural habitats in the tropics are well understood (Donald, 2004; Scharlemann *et al.,* 2004). However agricultural activities also occur at varying intensities, which then have varying levels of impact on land cover change and biodiversity loss (Donald *et al.,* 2001; Green *et al.,* 2005; Reidsma *et al.,* 2006; Buchanan *et al., 2009*). For example, the area of cropland has increased globally between 4.5 and 5 times from 1700 to 1990 (i.e. from an estimated 300–400 million ha in 1700 to 1500–1800 million ha in 1990 and a 50% net increase just in the twentieth century (Lambin *et al.,* 2003). There is therefore need to investigate and document the impact of the varying levels of agricultural intensities adjacent key biodiversity sites on land cover change within these sites in order to adjust conservation approaches, planning and make better and more effective choices.

1.1.4. Interventions at various scales

Key biodiversity sites such as Important Bird Areas (IBAs) have experienced adverse land cover change with a concomitant impact on species. IBAs are sites of international significance for the conservation of birds and other biodiversity recognized worldwide as practical tools for conservation. IBAs are identified using internationally agreed, standard, objective, quantitative and scientifically defensible criteria applied with common sense. They are distinct areas amenable for practical conservation and are part of a wider, integrated approach to conservation and sustainable use that embraces sites, species, habitats, and people. They are selected because they hold (a) bird species that are threatened with extinction (b) birds that have highly restricted distributions (c) species assemblages' characteristic of particular biomes and/ or (d) exceptionally large numbers of congregatory bird species (Fishpool and Evans, 2001).

The loss in land cover and biodiversity has attracted much attention as demonstrated by various efforts at local, national, regional and global levels to reverse current trends in deleterious land cover change and biodiversity loss. For example, primary conservation interventions include the designation of comprehensive networks of protected areas, species management, environmental education and awareness and law and policy interventions (Brooks, et al., 2009). As part of site-based land and water conservation interventions, worldwide designation of protected areas has been considered a very important mechanism to protect biodiversity and prevent biodiversity loss from deleterious human activities. As a result, there exists a very comprehensive network of 150,000 protected areas worldwide covering a total of 12.7% of the world's land area and 7.2% of its coastal waters sea (UN Statistics Division, 2011) and international forums have set targets for the global coverage of protected areas with respect to the global land surface (IUCN, 1993) which has been revised and agreed by the State Parties of the Convention on Biological Diversity to increase from 12% target set in 1992 to 17 % under COP10 by 2020 (CBD, 2010). However, despite this extent and comprehensive coverage of protected areas biodiversity is still declining due to inadequate management of existing sites and gaps in coverage (UN Statistics Division, 2011). Protected areas are designated to prevent biodiversity loss, yet there is considerable uncertainty and controversy over the impacts and effectiveness of protected areas (Nelson and Chomitz, 2009) and very few well designed evaluations (Ferraro and Pattanayak, 2006). A few assessments have been conducted focusing on certain habitat types (e.g. deforestation, Nepstad *et al.*, 2006; Brooks *et al.*, 2009; Nelson and Chomitz, 2009). Assessing the impact and effectiveness of site protection in preventing deleterious land cover change therefore, though challenging (Nepstad *et al.,* 2006) is critical in biodiversity conservation and achieving conservation targets set at various levels.

A series of international conventions and Multilateral Environmental Agreements (MEAs) and commitments for actions have been initiated and agreed upon with the ultimate goal being to significantly minimise current rates of biodiversity loss. All these conservation efforts also contribute directly or indirectly into global targets (e.g. Millennium Development Goals) and at national levels, countries are making all the necessary efforts in developing and domesticating Convention of Biological Diversity objectives and goals through National Biodiversity Strategies and Action Plans (NBSAPs). NBSAPs are the principal instruments for implementing the Convention at the national level with the key goal being to develop national strategies, plans or programmes for the conservation and sustainable use of biological diversity and over 173 State Parties have developed their NBSAPs (CBD, 2010). Governmental, non-governmental, private and civil society organisations and agencies are striving to conserve biodiversity through various interventions.

Countries such as Ecuador, Kenya, the Philippines, and Ukraine have been involved in initiatives to generate key biodiversity indicators to address key questions raised at national level and hence changes in land cover and land use and its effect on biodiversity was one of the key questions of concern (Bubb *et al.,* 2005).

All these national and global commitments, targets, efforts and interventions at the various levels require defined measurable indicators, attributes or conditions and systems to ensure timely and efficient information flow. Without systems to guarantee biodiversity information flow, it is difficult to realise key components of long-term goals and targets, formulate policies, plan and prioritise appropriately where, how and when to intervene, and monitor and evaluate the impact of policies, legislations and interventions on biodiversity conservation and sustainable development at various hierarchical levels.

Site-based conservation may be appropriate for the conservation of around 80% of species (Boyd *et al.*, 2008) and are widely recognized as being important in tackling global biodiversity loss (Joppa *et al.*, 2008; Nagendra, 2008). As part of halting the trends in habitat loss, various organisations and actors have initiated site-based conservation activities focusing on sites, species, landscapes and people (Brooks *et al.*, 2006; Ibisch and Bertzky, 2006).

The conservation interventions range from short to long-term projects and programmes involving local communities and working with various agencies on specific or integrated biodiversity and livelihood improvement interventions. There is very little rigorous evaluation of the impact of protected areas on local livelihoods (Ferraro and Pattanayak, 2006). However, there are also few assessments that demonstrate the impact of site based conservation interventions (e.g. working with communities). Roe *et al.*, (2009) report that a major deficiency of formal community-based natural resource management systems and programmes is the absence or paucity of quantitative and/or qualitative data on their social, economic and environmental impacts. There is therefore need for assessing the impact of local communities' conservation and natural resource management initiatives on reducing deleterious land cover change, which has a big impact on biodiversity at a wider scale.

1.1.5. Importance of data and information in conservation

Effective biodiversity conservation efforts depend on accurate, up-to-date and accessible information and data, efforts which according to Foody and Cutler (2006), are often limited by a lack of data. The information can be generated through analysing the state of biodiversity and assessing trends. This can be achieved through knowledge of 'state', and monitoring of the environment (e.g. Green *et al.,* 2005). Yet there exist a lot of gaps in terms of knowledge particularly on land cover change at key biodiversity sites.

Biodiversity monitoring is therefore central to pursuing current targets that focus on reducing biodiversity at various levels. Monitoring helps track land cover change and therefore monitoring of biodiversity is an essential component of conservation because it allows problems to be identified, priorities to be set, solutions to be developed, and resources to be targeted (Balmford *et al.,* 2003) and allows assessments of progress toward targets and indicators in unilateral and international conservation-policy instruments (e.g. CBD), impacts of international conservation policy (Donald *et al.,* 2007), effectiveness of site-based conservation interventions (Brooks *et al.,* 2009) and of other policy sectors (Donald *et al.,* 2001) and to inform conservation in order to adaptively manage ecosystems.

Monitoring requires defining indicators that are measurable. The Convention on Biological Diversity (CBD) adopted the state (quantity and quality of biodiversity), pressure (threats to biodiversity) and response (conservation efforts) indicators to measure progress towards achieving the 2010 target to reduce the rate of biodiversity loss (Balmford *et al.,* 2005) even though these targets were not met (Butchart *et al.,* 2010). However, their ability to plan and intervene is hampered by the paucity of comprehensive up-to-date information on spatial and temporal trends in biodiversity loss, the types and rates of land-cover and land-use change, and even less systematic evidence on the causes, distributions, rates, and consequences of those changes (Loveland *et al.,* 2002). For example, despite the significance of land cover change as an environmental variable, existing knowledge of land cover and its dynamics is very poor (Foody, 2002). Yet land cover change affects species and has the potential to drive certain species to extinction i.e. within Endemic Bird Areas (Scharlemann *et al.,* 2004). There also exist the Millennium Development Goals (MDG) targets and indicators, which place strong emphasis on eradication of poverty, improvement of lives and protection of the environment (UNDP, 2003).

The rate of habitat loss is increasing and the resources needed prioritise species and sites that are in critical need for conservation interventions. Resources for biodiversity conservation investments are also very scarce and therefore data from monitoring will provide the much needed information sites that are at most risk to facilitate timely intervention and efficient allocation of the scarce resources where they are needed most. Developing priorities requires information on sites, which are most at risk of deleterious land cover change. It is also possible to prioritise species that are in dire need of species focussed conservation intervention if the impact of land cover change on species is evaluated and those species most affected by these changes identified.

Further, monitoring can be used to assess the effectiveness of conservation interventions in terms of prevented or reduced rates of land cover change and increased conservation outcomes. The effectiveness of agri-environment schemes in Europe has been questioned (Kleijn *et al.,* 2006), and in the past couple of years there has been an increase in the understanding of the effectiveness of protected area designation (Dudley *et al.,* 2005). These studies help to evaluate how long-term investments have contributed to reducing current rates of biodiversity loss at site level, allowing re assessment of strategy where needed.

Conventionally, monitoring has been conducted through field based surveys, which involve field visits to collect data. However even though this approach is working well, the needed scale, level and coverage of the monitoring and taking advantage of recent revolution in alternative technologies become available, remote sensing and geographical information systems provides greater opportunities for environmental observation and conservation monitoring.

1.2. Statement of the Problem

Biodiversity is facing an extinction crisis as a result of the accelerated global land cover change and environmental degradation driven by both natural and anthropogenic factors. Yet the rate of decline in biodiversity is ever increasing and changes are threatening the survival of species and compromising the efforts being made to conserve biodiversity at various levels. Not only is data on land-use and land-cover change relatively poor, but there is need for a better understanding of the underlying driving forces for these changes. The need for understanding the spatial and temporal levels of land cover change, factors driving the change, the impact and cost-effectiveness of various forms of interventions and generating timely and accurate qualitative and quantitative information on status and trends of biodiversity and key biodiversity sites is critical if effective conservation is to be achieved.

Currently, the level and scale of monitoring is patchy and not only limited and restricted to a small subset of sites and habitat types (e.g. forests), but in some cases the monitoring is neither consistent, frequent nor systematic and the percentage and coverage of each site monitored is not always representative of the entire site and major habitat types of respective sites. There is therefore a need to increase the accuracy, coverage and frequency of habitat monitoring to ensure cost-effective and timely interventions are initiated to match the unprecedented increase in habitat loss. Resources are not adequate to generate as much information on as many sites as possible using ground-based methods. This coupled with logistical constraints necessitates the need to explore alternative and more cost-effective land cover and land use change analysis methods and techniques. The dramatic revolution in technological advancements makes it possible for these new systems and techniques to be explored, tried and adopted.

1.3. Justification of the Study

Filling the shortfall in information on the state of biodiversity, presenting up to date data on changes in habitats will allow conservation action to be better targeted, and resources used more efficiently. Further, understanding correlates of change will contribute to the identification of sites that may be at particularly high risk of degradation, or that require more frequent monitoring. It will also allow us to assess the impact of previous interventions on rates of land cover change. These data will be collected using a novel, low capacity approach to using satellite earth observation in monitoring key biodiversity sites.

The study is timely because with the current limitations associated with conventional ground-based IBA monitoring; there are still many sites of conservation importance, including many Protected Areas and other key biodiversity sites (e.g. IBAs) that do not benefit from any sort of systematic biodiversity monitoring (Hockings 2003; Green *et al.,* 2005). The implications of this poor coverage is that it may not be possible to evaluate conservation outcomes, keep track of the dynamics and trends in land cover (both positive and deleterious) at key sites for biodiversity conservation and identify the key drivers of habitat change at these sites. Earth observation data and associated tools therefore become a handy technique for use. The results from this study are therefore expected to improve and upscale the monitoring of IBAs for conservation and management, identifying priorities, targeting resources cost-effectively and ensuring timely interventions are made.

1.4. Objectives of the Study

i. To determine the percentage and extent of land cover change at IBAs using remotely sensed data;

ii. To describe correlates of actual habitat change on these IBAs that could be used to inform monitoring strategies, conservation management and advocacy;

iii. Based on i and ii, based on the first two key objectives, predict IBAs that are at risk of deleterious land cover change

iv. Develop methodologies and protocols for use of remote sensing in IBA monitoring;

1.5. Hypotheses

In the process of meeting the above objectives the following specific hypotheses will be tested:

a) Detrimental land cover change is higher at IBAs surrounded by dense human populations compared to change in sparsely populated IBAs;

b) Detrimental land cover change is greater at IBAs sorrounded by high agricultral intensity;

c) Detrimental land cover change is greater at unprotected IBAs;

d) Detrimental land cover change is less frequent following long-term site-based conservation initiatives.

e) Based on the above (a-d), it is possible to predict IBAs that are at risk of deleterious land cover change

1.6. Scope and limitations of the Study

This study examines the efficacy of earth observation data and remote sensing techniques in the monitoring of key biodiversity sites specifically Important Bird Areas. IBAs are defined in the context of BirdLife International (Fishpool and Evans, 2001). With over 1,230 IBA sites in Africa, the study is based on a small subset of 71 sites in eastern Africa. Even these 71 sites have been selected based on broad habitat types (e.g. forests, grasslands, shrub lands, wetlands and transitional habitats) geographical spread (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda) and availability of data.

Remote sensing techniques employed in the study include change detection analyses, which enabled spatial and temporal land cover change to be determined based on satellite data. Complementary ground-truthed data allowed verification and validation of the results from land cover change and determination of levels of image classification and interpretation accuracy between information on the satellite image and the actual information on the ground. Other datasets (hereby referred to as correlates of land cover and land use change) have been used in this study including human population density, agricultural intensity, protected areas and Site Support Groups (SSGs). While cognisant of the presence of other forms of threats or factors driving land cover and land use change, this study has only confined itself to human population density and agricultural intensity. In terms of site-based conservation interventions, even though these may be diverse, the study focuses only on two, namely; protection and working with communities. The study avoids delving into the broad subject of land use but instead focuses on land cover, which is basically the observed biophysical cover of the earth's surface captured in the distribution of vegetation, water immediate subsurface, including biota and those structures created solely by human activities (e.g. settlement, Lambin *et al.,* 2003). On the other hand, in the context of this study, land use is based on the functional dimension of land for different human purposes or economic activities such that intended employment and management strategy placed on land cover by human agents to exploit the land cover and reflects human activities such as residential zones, agricultural fields among others (Chrysoulakis *et al.*, 2004). Land use therefore affects land cover.

Statistical analyses were used in determining the relationships between the observed land cover change and the correlates of land cover change. Specifically, the study looks at two pressure correlates of land cover change at IBAs with particular focus on human population density and agricultural intensity. The study investigates the relationships between the rates and extent of detrimental land cover change as a result of the human population density and agricultural intensity around IBAs. A conservation intervention can be seen to be effective if it is making a real impact on biodiversity such that it reduces the threats and pressures driving detrimental land cover change at the focal sites and as a surrogate, reduces the risk of species becoming extinct.

Therefore to evaluate the impact of these interventions on IBAs, this study investigated whether land cover change is less pronounced at IBAs by focusing on two conservation interventions namely; protection and the establishment of SSGs. The rates of land cover change are correlated to these forms of interventions. Based on the land cover change analysis, potential impacts on species population and conservation status particularly those IBA trigger species that are restricted to specific habitat types are described. The data for all these correlates was available thus making these analyses possible. Amalgamating all these cause and effect factors enabled IBAs most at risk of deleterious land cover change to be identified. The study addresses fundamental issues associated with the dearth of information on impacts of pressures correlates (human population density and agricultural intensity) and conservation interventions (protected areas and community-based natural resource management) on land cover change at key biodiversity sites.

The overall goal of this study is to show how understanding the extent of detrimental land cover change at IBAs subjected to certain pressures (population density and agricultural intensity) is critical. The study also demonstrates how the trends in land cover changes following legal protection and long-term site-based conservation initiatives can aid efforts to evaluate effectiveness of interventions and identify habitats and key biodiversity sites that are more vulnerable and therefore in need of urgent conservation interventions to maximise conservation outcomes. The breadth of the thesis is therefore to demonstrate how remote sensing technique can be a cost-effective means of monitoring key biodiversity sites in combination with ground-based methods and hence vouches for its adoption and increased application within the entire conservation fraternity eventually addressing current management and monitoring problems.

1.7. Organisation of the Dissertation

This thesis is divided into five successive chapters and each focuses on a specific theme or subject. The introductory chapter provides an overview of the study in terms of issues affecting biodiversity, presents the statement of the problem, objectives of the study including the hypotheses to be tested, justification and the scope of the study. The subsequent four chapters focus on literature review, methodology, results and discussion and conclusions and recommendations respectively. The third chapter presents the methodology applied in the study by providing a description of the study area, data sources, data analysis methods and tools used in this study. The fourth chapter presents the results and discussion focusing on six study questions. Each of the study question subsections presents the introduction, the aims relevant to the chapter, methods, analysis, results and a discussion of the results. A general discussion of the overall results is made at the end of the chapter. The fifth chapter presents the conclusions, recommendations including areas that need action and further studies and finally the contribution of this study to the furtherance of science, particularly highlighting how the findings can be applied in the field of biodiversity conservation.

CHAPTER TWO: LITERATURE REVIEW

2.1. Biodiversity and Biodiversity Monitoring

2.1.1. Africa and biodiversity

According to UNEP (2007), in terms of species diversity, 25% (1,229 species) of the world's approximately 4,700 mammal species occur in Africa. Some 23% (2,313) of the world's 9,917 bird species are found in Africa. About 1,400 of these bird species are endemic to Africa. Africa and its adjacent islands has about 950 amphibian species while mainland Africa has between 40,000 and 60,000 plant species, of which approximately 35,000 are endemic.

2.1.2. Land cover change as a threat to biodiversity

Land cover changes are not simple processes but complex simultaneous patterns of land cover change, ranging from modifications in land cover to conversions (Lambin and Geist, 2006). Land cover change is critically linked to the intersection of natural and human influences in environmental change (Ghosh *et al.,* 1996) and these changes are driven by heterogeneous changes in land use (Turner *et al.,* 1995). For example, changes in patterns of human settlement and economic activities directly or indirectly cause land cover changes, whose pace has dramatically increased in the past few decades making land cover and land use change a major driver of global change (Turner *et al.,* 1993). Deleterious change in land cover is of major concern to conservationists because of its generally negative impact on biodiversity (Brooks *et al.,* 2002) especially considering that land use change is expected to be the primary driver of population reduction and species loss in the foreseeable future (Vitousek, 1994). Management and transformation of earth's surface through land cover and land use changes has been considered as one of the most significant global challenges this century (Mustard *et al.*, 2004).

For example, according to UNEP (2008), Africa's land resources are rapidly changing, and in some cases shrinking, due to changes in land cover, land use, and land productivity. In the forest habitat, between 1990 and 1997, 5.8 ± 1.4 million hectares of humid tropical forest were lost each year, with a further 2.3 ± 0.7 million hectares of forest visibly degraded (Achard *et al.*, 2002). Based on sub-Saharan level study (Brink and Eva, 2009), natural vegetation decreased by 21% over a 25 year period, with nearly 5 million hectares forest and non-forest natural vegetation and a 57% increase in agriculture area.

Land cover change analysis is important in identifying proportions of land cover categories that have been altered within a spatial and temporal framework. Such analysis and mapping of change have contributed to our understanding the dynamics and implications of human activities in time and space (Balmford *et al.*, 2003). Even though land cover change is happening across all major habitats, forest loss through deforestation is among the best-known examples of the largest threats to biodiversity and also the best studied and quantified example of land cover change. According to FAO (2005), Africa suffered the second largest net loss in forests with 4.0 million hectares cleared annually while leading in forest fires in the world. Further, approximately half of Africa's terrestrial ecoregions have lost more than 50% of their area to cultivation, degradation and urbanisation, (Burgess *et al.*, 2005) and nine other ecoregions lost more than 80% of their natural habitat (UNEP, 2006). The tropical forests where 50% to 70% of world's species are found (Whitmore, 1990) are lost at a rate of 0.5% to 2% a year with one fifth of the entire tropical rainforest cover lost between 1960 and 1990 (Mayaux *et al.*, 2005). Destruction and fragmentation of natural habitats are the two most important factors in the recent species extinction event (Groombridge, 1992). Primary in all this is expanding agriculture, which destroys more habitat than any other factor (FAO, 2001; Lambin *et al.*, 2001) and is predicted to be greatest in areas of high bird endemism (Scharlemann, *et al.*, 2005).

Continuing expansion of human population will mean that clearance for agriculture will continue, and perhaps increase (Tilman *et al.*, 2001). These land cover changes and the threats affect biodiversity at individual species level, habitat area and condition as well as ecosystem functions and services.

Habitat loss has already had an observed and predicted negative impact on species and global biodiversity at large (e.g. Myers *et al.*,2000; Warren, 2001; Brooks, *et al.*,2002, BirdLife International, 2004; 2008), and has influence beyond the direct loss, through, for example, fragmentation (Foin *et al.*,1998; Kruess and Tscharntke 1994; Gascon and Lovejoy 1998; Carvalho and Vasconcelos 1999; Hargis *et al.*,1999; Jansen *et al.*1999; Temple *et al.*,1999; Vickery and Gill, 1999; Vickery *et al.*,1999; Kurki *et al.*,2000; Laurance *et al.*,2000, Liu *et al.*,2001; Fahrig, 2002; Armenteras *et al.*,2003). Already, some 12 % of birds are classified as being globally threatened, with a further 9 % near threatened, while in Africa alone, of the 2,313 bird species, 235 are threatened within extinction (BirdLife International 2008). Other studies have investigated habitat loss as a driver of species loss (Brooks *et al.*, 2002). Considering the role of species in ecosystem function, the loss of 50% to 80% of the species of an ecosystem has the potential to cause the collapse of most biogeochemical processes and various components of ecosystem services (Cardinale *et al.*,2006; Worm, *et al.*, 2006; Ostfeld and Logiudice, 2003; Chapin *et al.*,2000). Understanding land cover change through conversion and modification can help improve conservation and ensure prompt interventions to reduce the impact of deleterious land cover change to species, sites, habitats and ecosystems and to develop sustainable land use.

2.1.3. Setting priorities and site-based conservation initiatives

The rapid destruction of habitats, increase in threats and decline in species requires the formulation, development and adoption of appropriate strategies, approaches and mechanisms to reduce the rate of biodiversity loss. Species diversity and endemism makes Africa very important for biodiversity conservation and therefore prioritising sites for conservation action is critical especially in view of current rates of habitat loss, increasing human population density and associated demands and pressure on natural resources.

Based on inherent interest by institutions, many prioritisation schemes and methods have been applied at different spatial scales, including those based on single-taxon criteria (Pain *et al.*, 2005). A growing number of global initiatives have identified areas of high conservation value or importance at regional scales (Brooks *et al.*, 2006) and at scales amenable to management. Various conservation agencies and multilateral agreements working with State Parties have identified broad, extensive areas that are of highest conservation priority across the globe based on varying criteria. Although there is often lack of concordance between different site selections processes (Brooks *et al.*, 2006), there is a lot of complementarity and overlaps amongst these areas in many instances. For example, the majority of IBAs in Africa (57% of the 1230 sites) overlap to varying degrees with some protected area (BirdLife International, 2005).

African biodiversity is not evenly distributed across the continent, as has been recognised by several 'prioritisation' initiatives (Table 1). For example, there are 40 Endemic Bird Areas in Africa (Stattersfield *et al.*, 1998), while eight of the world's 34 biodiversity hotspots have been designated in Africa (Mittermeier, *et al.*, 2004). These broad areas are very extensive to be amenable to focused management and conservation initiatives already in place. Instead, further site based initiatives have identified areas that are amenable to management, including the Protected area network (IUCN and UNEP 2008), whose sites form a pillar of global conservation strategy, but are not identified based on any objective criteria (Chape *et al.*,2003).

The protected area network is a widely recognised approach for protecting global biodiversity, maintaining environmental services, and protecting cultural, aesthetic, and ethical values (Hockings *et al.*, 2000). In order to conserve, protect and maintain condition and integrity of ecosystems and biodiversity, protected areas have been designated across the globe and extended in some cases. With 12% of the global surface under protection, this surpasses the 10% of the global surface under protected area status target (IUCN 2005); this demonstrates how this intervention strategy has been fervently pursued at local, national, regional and global levels. The extent of coverage is due to increase to 17% (CDB). Africa contains over 6,456 of the 161,991 global network of protected areas (IUCN and UNEP-WCMC, 2010) covering more than 7% (2million km^2) of the continent's land area. . However, this network delivers relatively poor coverage for species (Rodrigues *et al.*, 2001), especially those of highest conservation importance (Beresford *et al.*, 2012, *In Prep0*). Certain areas and regions have been recognised in their own right by various organisations as meeting the respective criteria used in their identification but with some overlap among these classifications as well as lack of concordance between different site selection processes to some degree (Brooks *et al.*, 2006). However, protected areas are generally not identified in any objective way. Identifying such areas provides valuable guidance in targeting conservation efforts (Statersfield *et al.*, 1998).

By contrast, the global network of Important Bird Areas is a priority setting approach identified and adopted based on objective criteria by BirdLife International[1] as part of its global conservation strategy, and areas are of a size and nature amenable to management. IBAs are thus sites of international significance for the conservation of birds and indeed other biodiversity (Pain *et al.*,2005), recognised worldwide as practical tools for conservation and form an integrated approach to conservation and sustainable use that embraces sites, species, habitats, and people (Fishpool and Evans 2001). IBAs are selected because they hold bird species that are threatened with extinction, have highly restricted distributions, congregations or are characteristic of particular biomes (BirdLife International, 2005) as per the criteria in Annex 1. The identification of IBAs is the first step towards their conservation and the subsequent monitoring of their integrity and the bird populations within them is a high priority (Bennun, 2002). IBAs overlap extensively with the protected areas (Fishpool and Evans 2001), but differ in that IBA sites are selected based on objective criteria.

[1] BirdLife International is a global partnership of 117 national, civil society conservation organizations striving to conserve birds, their habitats and global biodiversity, working with people towards the sustainable use of natural resources. In Africa, the BirdLife Partnership is a growing network of 22 such organizations plus one Country Programme covering 23 countries, with a combined total of more than 300 staff and 30,000 members.

Table 1: Conservation prioritisations as identified and designated by respective organisations

Prioritisation	Criteria	Coverage	Reference	Organisation
Biodiversity hotspots	Areas of particularly high species richness and endemism, and under particular threat	34 globally, 8 in Africa	Mittermeier *et al.*, 2004	Conservation International
Centres of Plant Diversity	Most important sites for plants worldwide that are priority areas for conservation.	global	Davis *et al.*,1994	WWF, IUCN
Important Bird Areas	Sites of international significance for the conservation of birds and other biodiversity selected because they hold (a) bird species that are threatened with extinction (b)birds that have highly restricted distributions (c)species assemblages characteristic of particular biomes and/ or (d)exceptionally large numbers of congregatory bird species	11,000 globally, 1,230 in Africa	Fishpool and Evans 2001	BirdLife International
Endemic Bird Areas	An area which encompasses the overlapping breeding ranges of restricted-range species, such that the complete ranges of two or more range-restricted species are entirely included within the boundary of the EBA	218 EBAs globally, 37EBAs, 27 Secondary areas in Africa	ICBP, 1992; Stattersfield *et al.*,1998	BirdLife International
African Heartlands	Cohesive conservation landscapes that are biologically important and cover areas large	9 heartlands in east, south and west Africa	http://www.awf.org/section/heartlan	African Wildlife Foundation

	enough to maintain healthy populations of wild species and natural processes well into the future		ds	
Ecoregions	a large area of land or water that contains a geographically distinct assemblage of natural communities that (a) share a large majority of their species and ecological dynamics; (b) share similar environmental conditions, and; (c) interact ecologically in ways that are critical for their long-term persistence.	825 terrestrial and 426 freshwater ecoregions delineated globally	Olson and Dinerstein, 1998; Olson *et al.*, 2001,	WWF
Key Biodiversity Areas	Places of international importance for the conservation of biodiversity through protected areas and other governance mechanisms. They are identified nationally using simple, standard criteria, based on their importance in maintaining populations of species.	Of the 34 biodiversity hotspots, 8 are in Africa	http://www.globalconservation.info	Conservation International/BirdLife International
Frontier Forests	Are areas of pristine habitat where natural processes may be most readily maintained	Global	Bryant *et al.*, 1997	The World Resources Institute
Protected Areas	Area of land and/or sea especially dedicated to the protection and maintenance of biological diversity and of	More than 1,254 Protected Areas designated in Africa	IUCN 1994	IUCN and World Commission on Protected Areas

	natural and associated cultural resources, managed through legal or other effective means. Also a geographically defined area which is designated or regulated and managed to achieve specific conservation objectives		Chape *et al.*, 2003 ; Chape *et al.*, 2005	CBD
Wetlands of International Importance	List of Wetlands of International Importance selected because they contain representative, rare or unique wetland types and are of international importance for conserving biological diversity	Sites designated in the world: 1,886 covering total surface area 185,156,612ha	http://www.ramsar.org	Ramsar Convention
World Heritage Sites	Sites of outstanding universal value in accordance with the UNESCO Convention on the Protection of the World Cultural and Natural Heritage (1972) and are nominated by the State Party	689 sites nominated globally, 78 cultural, natural or mixed World Heritage Sites nominated in Africa	http://whc.unesco.org/en/list/	UNESCO World Heritage Convention
Man and Biosphere Reserves	Areas of terrestrial and coastal ecosystems promoting solutions to reconcile the conservation of biodiversity with its sustainable use, are internationally recognised, nominated by national governments and remain under sovereign jurisdiction of the states where they are located. They serve in	Over 500 Biosphere Reserves in 100 countries,	http://portal.unesco.org/science	UNESCO's Man and the Biosphere Programme

	some ways as 'living laboratories' for testing out and demonstrating integrated management of land, water and biodiversity.			

A network of over 1,230 Important Bird Areas covering 2 million km^2 (7% of the region's land area) has been identified across the continent (Figure 1) and this information is contained in a regional IBA directory (Fishpool and Evans, 2001). Following the identification of these sites, based on objective criteria, it is important that their habitat condition, quality, integrity and ecological processes are monitored (Fishpool and Evans, 2001; Bennun, 2002).

The IBA conservation programme involves setting priorities and implementing advocacy, monitoring and actions for key sites and developing national site conservation programmes. IBAs therefore become priority sites for conservation action, research and monitoring and integrating community-based natural resource management through livelihood improvement initiatives. Hence, BirdLife International endeavours to establish a sustainable management cycle in which a programme of action, monitoring and advocacy for the network of national IBAs is well established, with security of future funding. This programme is supported by local Site Support Groups (SSGs)[2] that are focused on IBAs. These can deliver useful information for site monitoring (Mwangi *et al.,* 2010), and by 2010, 211 Site Support Groups have been established to contribute to the management and monitoring of over 117 IBAs across the continent (BirdLife International, 2010). This is an important element of BirdLife work as a model, and it entails

[2] Site Support Groups (also referred to Local Conservation Groups, LCGs) are organized groups of voluntary individuals who work, in partnership with relevant stakeholders, to promote conservation and sustainable development at IBAs and other key biodiversity sites

Community-Based Natural Resource Management where communities living around key biodiversity sites become involved in the management and conservation of these sites through an integrated system that includes conservation, monitoring, capacity building and livelihood improvement.

At key sites across the network, BirdLife International and its network of Partners are implementing a range of biodiversity conservation and poverty alleviation initiatives. Site-based conservation initiatives include rehabilitation and restoration of degraded sites, education and awareness raising, policing and law enforcement, boundary demarcation, research and monitoring. Monitoring the status and trends of biodiversity and tracking the impact of conservation interventions at these sites have been identified as priorities for conservation (BirdLife International, 2006).

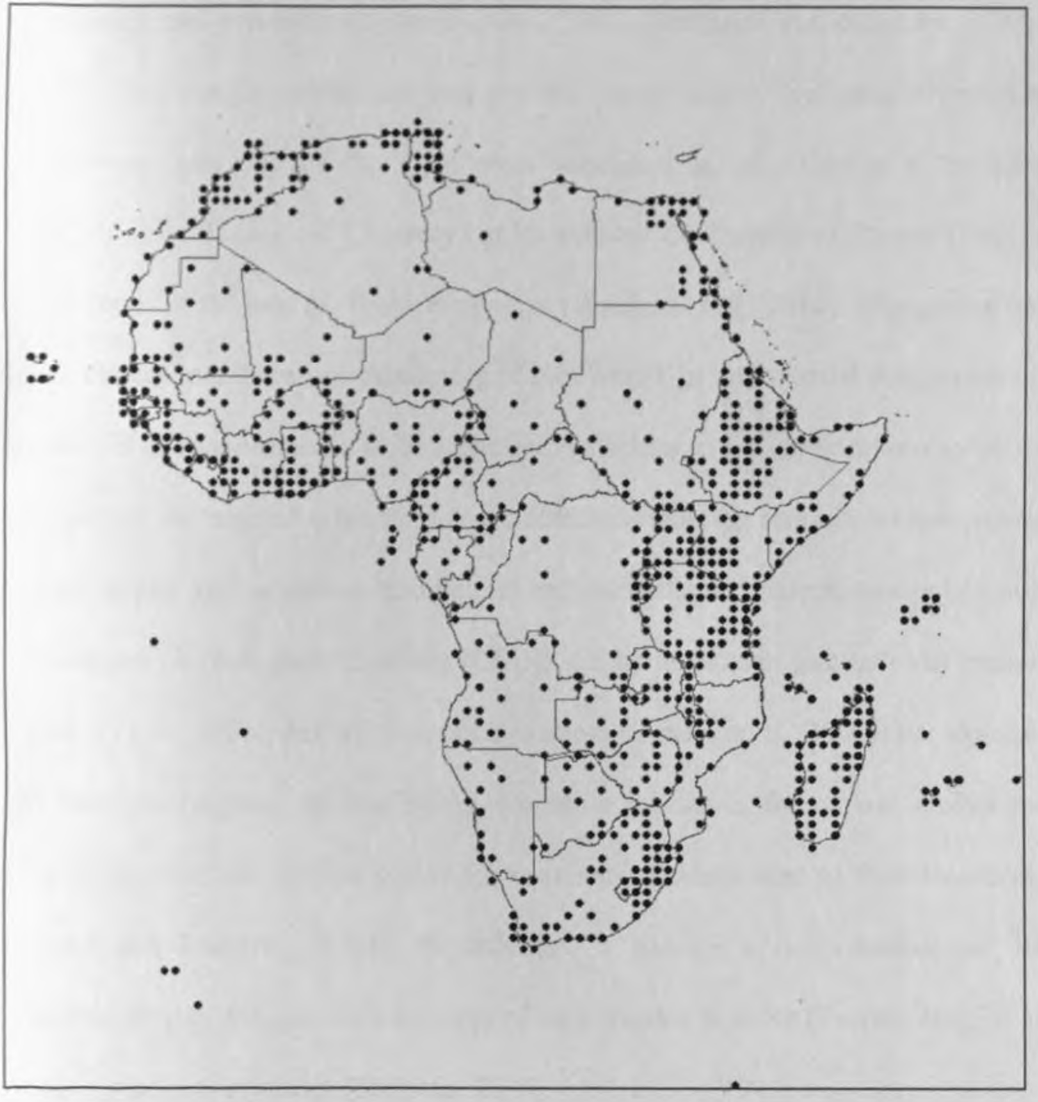

Figure 1 Over 1230 IBAs have been identified in mainland Africa and adjacent islands (Source: BirdLife International)

2.1.4. What is biodiversity monitoring and why monitor?

The need to track spatial and temporal changes in land cover is now greater than before (Kerr and Ostrovsky 2003; Buchanan *et al.,* 2009b). The rate at which natural resources continue to be degraded has caused much concern at local, national and international levels (Millennium Ecosystem Assessment, 2005) leading to greater scrutiny and development of effective monitoring and evaluation systems (Noss, 1990; Margoluis and Salafsky, 1998; Teder *et al.*, 2007). For example, status and trends of the components of biological diversity and threats to biodiversity was one of the focal areas identified as an indicator to be adopted by the Convention of Biological Diversity at its seventh Conference of Parties (Decision VII/30) to track trends in the rate of biodiversity loss (Butchart *et al.*, 2010). Monitoring helps track land cover change and therefore monitoring of biodiversity is an essential component of conservation because it allows problems to be identified, priorities to be set, solutions to be developed, and resources to be targeted (Balmford *et al.*, 2003). Monitoring also allows assessments of progress toward targets and indicators in unilateral and international conservation-policy instruments (e.g. Convention on Biological Diversity [CDB]), of the impacts of international conservation policy (Donald *et al.*, 2007), and of other policy sectors (Donald *et al.*, 2001). For example, monitoring has been instrumental in reinforcing economic incentives for carbon credits for forests and supporting countries receive credits for sustainable management of their forests as carbon sinks (Cairns and Lasserre, 2004). Nevertheless, a paucity of information has led to a poor understanding of the cost-effectiveness of conservation policies (Ferraro and Pattanayak, 2006), exposing them to criticism (Stokstad, 2005).

Effective biodiversity conservation depends on accurate, up-to-date and accessible information. Biodiversity monitoring can be complex especially given the diversity of actors involved, plus the diversity of aspects to be monitored, varying from species level (and the high species diversity), to site, habitat, landscape and ecosystem levels. Whereas the need to improve monitoring is widely recognised (Balmford *et al.*, 2003), the need to develop monitoring systems and protocols that are simple, sustainable, and readily applicable to a range of habitat types and threats is recognised (Buchanan, *et al.*, 2009b).

Global conservation needs far exceed the available resources; so scarce resources need to be used cost-effectively to maximise returns on conservation investment. (Balmford *et al.,* 2003; Murdoch *et al.,* 2007). Setting priorities and allocating resources require first-hand information on the nature and levels of threats (Margules and Pressley, 2000), which threats are increasing, have more impact, where they are greatest and how they will change in future. It helps to identify which actors or underlying drivers to target as part of focused interventions. Monitoring enables identification and geographical determination of locations where changes in loss or gain in land cover has happened or conservation interventions are needed most (Feoli *et al.*,2002; Vasconcelos *et al.*,2002; Ayyad, 2003; Zhao *et al.*,2003) hence being critical in the priority setting process, ultimately ensuring effective and efficient allocation of scarce resources. It also makes it possible to detect and act on threats in good time and provide up-to-date spatial and temporal information on biodiversity trends enabling the design and implementation of appropriate conservation strategies. For example, vegetation monitoring is considered an important component of most restoration monitoring programs (Phinn *et al.,* 1996). It is also possible to determine the drivers and underlying factors to land cover change. In conservation, this forms a very crucial stage of adaptive management; an iterative process, which enables managers and scientists to improve resource management over time by learning from management outcomes and using the results to update knowledge and adjust future management interventions accordingly. Targeting priority species and sites of highest conservation value such as IBAs, KBAs and others (e.g. Brooks *et al.,* 2006) would result in the highest biodiversity conservation priorities being monitored.

Depending on the situation and the necessity, monitoring can be tailor-made to track progress, evaluate and quantify the impact of past conservation and management actions at species and site levels and on people's livelihoods. At the global level, monitoring of the extent of habitat loss and transformation could also inform and influence decisions and feed into conventions and national reporting mechanisms to Multilateral Environmental Agreements and Conventions such as Convention of Biological Diversity. Target conventions include Convention on Biological Diversity's 2010 targets to reduce biodiversity loss (United Nations, 2002), progress towards the Millennium Development Goals, Ramsar Convention on Wetlands of International Importance, Convention on Migratory Species (CMS), African-Eurasian Waterbirds Agreement (AEWA), United Nations Convention to Combat Desertification (UNCCD), UN Framework Convention on Climatic Change (UNFCCC), UNESCO World Heritage Convention among other global targets. Most of these international agreements have been developed as part of the concern for nature and unprecedented loss of biodiversity and commitment and guide in conservation and ecosystem management.

However, monitoring of biodiversity trends and outcomes is one area that has been particularly challenging to global conservation (Green *et al.,* 2005). The ability of governments and capacity of organisations to effectively monitor vast territories in order to manage environmental change for sustainable development is limited. For example, in many countries, it has proved challenging to monitor accurately the extent of humid tropical deforestation due to poor monitoring capacity, infrastructure and inconsistencies among monitoring regimes (Kerr and Ostrovsky, 2003). This is in spite of forest loss through deforestation being among the best-known examples of the largest threats to biodiversity and also the best studied and quantified example of land cover change. This makes monitoring a challenging task for conservation practitioners at the local, national, regional and global levels and particularly so in the developing world where capacity and financial resources are scarce. Even for BirdLife International, only around 200 IBA sites are being monitored out of the 1,230 sites in Africa (BirdLife International 2010). This monitoring generates very useful information for policy and advocacy. However, it is not regionally representative of the entire diversity of sites on the continent and does not include the countries where such information on status and trends of sites would be useful for setting priorities to reduce the unprecedented increase in detrimental land cover change and the concomitant loss in species.

2.1.5. Monitoring frameworks

Significant efforts by various actors have been invested in monitoring biodiversity as evidenced by a plethora of diverse tools and indicators, methods, frameworks and approaches developed and being applied by various institutions and practitioners to effectively monitor biodiversity at various hierarchical levels. Although variable, these monitoring tools play complementary roles with varying degrees of overlap, applications, coverage, focus, strengths and weakness and hence preclude or promote their use by the various users. For example, some of the commonly used monitoring indicators and tools in the Eastern Arc and Coastal Forests of Kenya and Tanzania and their levels of application are shown in Table 2.

Table 2: List of various monitoring indicators and tools being used in the Eastern Arc Mountains and Coastal Forests of Kenya and Tanzania.

Indicator		Level at which it is being applied	Main tools/methods
State Indicators	Forest quality and forest health	Site/habitat	• Disturbance Transects • IBA Monitoring Framework • Remote Sensing (aerial surveys and image analysis)
	Area of different types of forest and degree of fragmentation	Site/habitat	• Remote Sensing (aerial surveys and image analysis) • Habitat characterisation and ground-truthing • Patch analysis
	Presence of endemic and globally threatened species	Site/habitat	• Methods will vary with the taxa selected
	Change in species IUCN Red List Category (Vulnerable, Endangered, Critically Endangered, etc.)	Species	• IUCN Red List Index • Data Analysis
	Change in species abundance for a few key species (e.g. endemics, threatened, migratory, or other 'flagship' species)	Species	• Field Surveys • IBA Monitoring Framework
	Forest Cover change	Landscape Site/habitat	• Remote Sensing (aerial surveys and image analysis) • Forest Cover Change • Forest Health Monitoring Framework
	Gaps in a) national legal recognition; b) international acceptance of nationally legislated reserves; c) making biodiversity conservation an official goal of key biodiversity areas.	Site/habitat	• GIS • Evaluating gazettement list • Questionnaire with site managers • IBA Monitoring Framework • Site Surveys
	Percentage area within Protected Areas	Landscape Site/habitat	• Maps • GIS • World Database on Protected Areas
	Environmental (ecological and economic) services from the site e.g. quality and quantity of water flowing from the site, soil erosion, non-timber forest products, pollination	Site/habitat	• Hydrological surveys • Soil erosion measurements • Economic valuation and PRA

Pressure Indicators	Change in extraction intensity of globally threatened species for commercial use	Species	• Data derived from TRAFFIC database and Disturbance transects
	Change in human population density in wards/divisions containing Eastern Arc Mountains and Coastal Forests of Kenya and Tanzania	Landscape	• Review of National Bureaus of Statistics Reports in Kenya and Tanzania • Gridded Population of the World database • GIS
	Fire Frequency	Landscape/Site/Habitat	• Remote sensing (MODIS fire points)
Response Indicators	Change in protection status of Key Biodiversity Areas (KBAs)	Site/Landscape	• Tracking the change in percentage of KBAs with official protection status using e.g. the World Database on Protected Areas (WDPA)
	Changes in forest management effectiveness	Site/Landscape	• METT (The World Bank/WWF Management Effectiveness Tracking Tool Indices)
	Presence and use of management Plan to protect threatened species	Species/Site/ Habitat	• Management Plans • IBA monitoring Framework
	Actions and research targeting key (threatened/endemic/migratory) species	Species	• IBA monitoring Framework • Survey of Research initiatives (i.e. number of research projects per year, number of publications per year, amount of funding allocated for research per year)
	Policy development (include site, species focussed issues)	Landscape/Site/Habitat/ Species	• Legal notices • Revised policies, laws, regulations
	Number of sites from which benefits accrue to local communities	Site/Habitat/Landscape	• Household questionnaires • Participatory Rural Appraisal • Rapid Rural Appraisal
	Change in policies/rules to reduce tourist practices with negative impact on threatened/endemic species	Species/Site/Habitat	• Surveys/assessment of tourism related policy • IBA monitoring Framework

Biodiversity is too complex in nature, and needs to be quantified at scales that are useful for the policy making process. To effectively monitor biodiversity, there is need for appropriate tools, methodologies, indicators and the need to define the geographical scale of monitoring (e.g. species, site, habitat, landscape, and ecosystem or biome levels). BirdLife International is applying the IBA monitoring framework within the context of the IBA programme. There exists a regional IBA framework (Bennun, 2002) and now a global IBA framework (BirdLife International, 2006), which as tools advocate and provide guidelines for monitoring IBAs across the region and the globe. Monitoring of a limited number of IBAs is ground-based and at two levels of detail; broad level (habitat extent and integrity), while on a limited number of IBAs, a more detailed level, i.e. monitoring of finer scale habitat characteristics and bird species presence and abundance (BirdLife International, 2005).

As with other monitoring schemes and frameworks, both BirdLife regional and global frameworks place greater emphasis on sustainability. This ensures the system is kept simple, robust, and inexpensive and involves management authority or project staff, Site Support Group members/IBA caretakers and other volunteers (BirdLife International, 2005; 2006). Site Support Group/IBA Local Conservation Groups are groups of volunteers who, in partnership with relevant stakeholders, help promote conservation and sustainable development at IBAs and other key biodiversity sites (BirdLife International, 2007a; 2010a). These groups play a crucial role in ground-based monitoring at a suite of IBAs in Africa. Currently about 140 Site Support Groups have been established to contribute to the management and monitoring of at least 150 IBAs.

2.1.6. Limitations of conventional monitoring methods

Conventional ground-based, community-led monitoring programmes continue to be valuable in engaging communities in both biodiversity and livelihood initiatives, which BirdLife International is implementing. However, even though it has been successful in some areas (e.g. generation of qualitative data and information, participatory nature of data collection, enhancement of management responses at local spatial scales and increased speed of decision-making (Danielsen *et al.*,2010), this conventional monitoring approach has its limitations (e.g. Green *et al.*,2005). Conventional monitoring through manual, field-based observation can generate accurate measurements but since it is labour intensive, is practical only for local-scale studies (Aplin, 2005). The approach has been demonstrated to be expensive, time consuming (Green *et al.*,1993; Barrett and Curtis, 1999), arduous and often requiring specialist skills and resources and is difficult in remote or politically insecure areas (Buchanan *et al.*,2009b). Moreover, the coverage of ground-based monitoring is limited by sites' remoteness, (Mwangi *et al.*, 2010) as epitomised by inaccessibility and rugged terrains and insecurity in some areas and is impractical for anything other than local-scale studies (Aplin 2005). According to Phinn *et al.*, (1996) ground-based sampling techniques may introduce disturbances into the area being sampled. This is especially true if disturbances are in the form of intrusive establishment of large transects in a habitat that may involve clearing of natural vegetation. In less wealthy countries, most monitoring programmes have also been noted to be unrealistically large, complicated and not sustainable after donor funding (Danida, 2000).

Limited access to data and information and lack of infrastructure exacerbates the problem and also limits progress with developing monitoring schemes (Ministry of Environment and Natural Resources 1994; 2000). In terms of its application for BirdLife programmes, the fact that BirdLife is currently working only in 23 countries in Africa and even then, on-the-ground monitoring is not taking place in all these countries (as a result of the reasons above) limits the spatial coverage and leaves the approach unsustainable. Complementary monitoring systems and approaches to ensure sustainability and robustness of the data and information are needed.

Consequently, alternative solutions need to be developed. The potential of remote sensing, in particular satellite imagery, for monitoring the environment has long been recognised (e.g. Tucker *et al.*, 1979), and this equally applies to biodiversity monitoring (e.g. Kerr and Ostrovsky, 2003; Pettorelli *et al.*, 2005). Already, it is applied to monitor forest cover change (Mayaux *et al.*, 2005, Hansen *et al.*,2009), but we still lack a system that exploits the potential for monitoring sites of conservation importance, despite some considering this now (Buchanan *et al.*,2009b). Satellite image based monitoring could be particularly useful for IBA monitoring since the most prevalent threats to IBAs in Africa involve land cover change that can be assessed and detected by satellite imagery (Buchanan *et al.*,2009a).

2.2. Remote sensing and Geographic Information Systems (GIS)

Remote sensing is the obtaining of information about features or phenomena from data acquired by a device that records reflected, emitted, or diffracted electromagnetic energy, and which is not in direct contact with the features or phenomena under investigation (Lillesand and Keifer, 2000; Jensen, 2000). It deals with the acquisition, processing and interpretation of images and related data obtained from aircrafts and satellites that record the interaction between matter and electromagnetic radiation. Earth observation satellites measure reflected, emitted or backscattered electromagnetic radiation from the earth's surface using instruments stationed at a distance from the site of interest. Satellite images are taken by space borne sensors that measure variation in light reflected from the surface of the earth; essentially, they take images from space, though unlike ordinary cameras, they collect information from wavelengths above and below those visible to the human eye. The radiation they capture can be that reflected by the Earth's surface from natural sources (passive sensors), such as sunlight, or that reflected by the same surface from an artificial radar source mounted on the satellite itself (active sensors). The characteristics of the surface of the Earth cause variation in the reflectance measured by the sensors. In the context of land cover, it is equivalent to surveying of land cover without physically visiting the site thus enabling monitoring and measuring of biophysical characteristics of the earth. Satellite images form a subset of this, and for the purpose of this review they are any space borne platforms that collect data on the surface of the earth. Satellite images can therefore be used to make objective assessments of land cover across areas that are too vast to survey on the ground, too inaccessible or too politically unstable. Applications of remote sensing through satellite imagery in conservation and monitoring land cover are widespread.

These remote sensing technologies provide data, information and tools for environmental practitioners to use to analyse and fully understand an ecosystem, allowing decision makers to better visualise, integrate, and quantify available natural resource data as well as ecosystem dynamics for better management and tailor-made interventions. The availability of historical and a wide range of spatial and temporal resolutions of remotely sensed data, the reduction in data cost and increased resolution from satellite platforms, remote sensing technology appears to make an even greater impact on systematic monitoring of land-cover and land-use change at a variety of spatial and temporal scales thus making remote sensing a valuable source of land-cover and land-use information (Rogan and Chen, 2004). Rapid technological advancements and improved sensor systems propelled remote sensing into a stage of exponential growth toward an era, where reliable information can be generated and shared routinely for various applications (Curran,1985) and this rapid advancement has been facilitated by substantial improvements in remote sensing methods and applications (Franklin, 2001). The costs of remote sensing techniques have previously been shown to be cost-effective in tropical environments (Mumby *et al.*, 1999).

Ecological applications require data from broad spatial extent beyond field-based capabilities. From early studies that have demonstrated links between bird distributions and reflectance (e.g. Nøhr and Jørgensen, 1997), and even earlier assessments of land cover change (Lambin *et al.*, 2003), thinking developed around the need to identify the potential of satellite imagery in biodiversity assessment (e.g., Skole and Tucker, 1993; Nagendra, 2001). An increasing number of publications and reviews have emphasized past uses of satellite images, and/or identified where they have the potential to improve biodiversity monitoring (e.g. Kerr and Ostrovsky 2003; Gottschalk *et al.*,2005, Pettorelli *et al.*,2005; Leyequien *et al.*,2007). Remote sensing analysis therefore offers the opportunity to keep track of species assemblages, habitats, ecosystems, ecological processes, impact or extents of catastrophic events such as fires and floods. It is able to help detect or monitor areas over time to determine local or large-scale natural and human-induced spatial environmental changes that drive landscape composition and configuration (Kerr and Ostrovsky, 2003).

Many studies that have used satellite images to map land cover or study biodiversity have used data from just one time point (e.g. Buchanan *et al.*, 2005). While these studies are useful, they do not describe changes in land cover, or its suitability for monitoring species. Satellite image analysis is particularly appropriate for land cover change and habitat loss analyses because satellites revisit the same location on the earth's surface at regular intervals (temporal resolution) and hence the ease of accessing biophysical and habitat data including ecological parameters such as habitat extent, heterogeneity or primary productivity. Data acquired from remote sensing techniques can provide comparable, spatially explicit information over large areas (larger than can be covered by field methods), decreasing the need for extrapolation from limited samples to large spatial scales where ground-based methods may fail to capture full extents or distributions (Shuman and Ambrose, 2003). Further, given that satellite data have been available continuously for over 30 years, it is possible to undertake retrospective analysis of change, developing baselines beyond those which could be used if based on conventional field work methods. The increasing emphasis of monitoring, development of new sensor technology, new techniques for applying remote sensing will increase the application and opportunities of remote sensing in research and biodiversity conservation.

Geographic Information System (GIS) has various definitions. GIS is a computer based information system that enables the input, management, manipulation, analysis, output and dissemination of all kinds of spatially referenced land related data and information at all scales (Mulaku, 2000). Burrough and McDonnell (2000) define it as a powerful set of tools for collecting, storing, retrieving at will, transforming and displaying spatial data from the real world for a particular set of purposes. Other definitions of GIS are presented in Table 3. However all these definitions have a functional point of convergence in terms of the fact that GIS provides a framework for gathering and organising spatial data and related information so that it can be analysed, displayed and used for decision making. According to Aronoff (2005), many applications implemented using GIS depend on datasets derived from remotely sensed data or make use of imagery directly as a background in graphic displays.

Table 3: Definitions of GIS (adapted from Burrough and McDonnell, 2000 with a few modifications)

Forms	**Definition**	**Reference**
Toolbox-based definitions	A powerful set of tools for collecting, storing, retrieving at will, transforming and displaying spatial data from the real world	Burrough, 1986
	A system for capturing, storing, checking, manipulating, analysing and displaying data which are spatially referenced to the Earth	Department of Environment, 1987
	An information technology which stores, analyses and displays both spatial and non-spatial data	Parker, 1988
Database definitions	A database system in which most of the data are spatially indexed, and upon which a set of procedures operated in order to answer queries about spatial entities in the database	Smith *et al.*,1987
	Any manual or computer based set of procedures used to store and manipulate geographically referenced data	Aronoff, 1989
	Is an integrated collection of computer software and data used to view and manage information about geographic places, analyses spatial relationships and model spatial processes	Aronoff, 2005
Database definitions	A data management, manipulation, analysis and display system based on sets of data that combine the characteristics of an object or feature and its geographical location. The datasets may be map oriented where features are represented in vector format, by lines, points and polygons or image oriented where features are represented in raster format as integer values cells in a rectangular grid	ESRI, 2006
Database definitions	A computer-based system that captures, stores, manages, analyses, and displays georeferenced data (geographic data)'. Many data relating to environmental and ecological systems have been collected and stored in forms suited to management and analysis using GIS	Aspinall, 1995

Organisation-based definitions	An automated set of functions that provides professionals with advanced capabilities for the storage, retrieval, manipulation and display of geographically located data	Ozemoy *et al.*, 1981
	An institutional entity, reflecting an organisational structure that integrates technology with a database, expertise and continuing financial support overtime	Carter, 1989
	A decision support system involving the integration of spatially referenced data in a problem solving environment	Cowen, 1988

GIS and remote sensing are complementary in many ways. Remote sensing serves as a major data input source for GIS. Spatial analysis which is the process of examining the locations, attributes and relationships of features in spatial data through overlay and other analytical techniques in order to address a question or gain useful knowledge is one key area of convergence. GIS therefore in combination with remote sensing has a very great potential application in capturing, storing, checking, manipulating, analysing and displaying IBA spatial data. This coupled with a revolution in the availability of information and in the development and application of tools for managing information makes the application of GIS and remote sensing in biodiversity information management and dissemination a reality.

2.3. Applications of Remote sensing and GIS in Biodiversity Conservation and Monitoring

The potential use of satellite images in biodiversity related studies is known because of its capability to provide a source of information on biodiversity at landscape, regional, continental and global spatial scales (e.g. Nagendra, 2001; Willis and Whittaker, 2002; Turner *et al.*,2003, Gottschalk *et al.*,2005; de Sherbinin, 2005). At a continental level, Buchanan *et al.*, (2011) reviews how satellite imagery can be used for African bird conservation. In fact its ability to offer an inexpensive means of deriving complete spatial coverage of environmental information for large areas in a consistent manner that may be updated regularly has been established (Muldavin *et al.*,2001; Duro *et al.*,2007). For example, remote sensing has the ability to produce spatially explicit information of large habitat areas, which can allow the detection of spatially explicit patterns that may not have been realised by field sampling techniques (Shuman and Ambrose, 2003).

Remote sensing data and techniques address these needs, which include identifying and detailing the biophysical characteristics of species' habitats, predicting the distribution of species and spatial variability in species richness, and detecting natural and human-caused change at scales ranging from individual landscapes to the entire world (Kerr and Ostrovsky, 2003). Remote sensing is increasingly used by policy-makers and conservationists to identify and map high priority conservation areas, map species distributions and diversity, examine land cover changes and identify those deleterious changes that could affect species (Buchanan *et al.,* 2008). Remote sensing is useful in ecosystem management treaties (Multilateral Environmental Agreements) and trans-boundary conservation. de Sherbinin, (2005) demonstrates the different ways in which remote sensing data and analyses can be applied in support of these conventions including land cover mapping, mapping species habitats and distribution. However, to maximise, catalyse and fully exploit the use of the full range of remote sensing techniques in ecology, land cover mapping, land cover change and conservation in general will require seamless integration of remote sensed data with existing and new ecological data (Kerr and Ostrovsky, 2003). In this review, examples of where the tool has been applied in ecology, ecosystem management, biodiversity conservation and monitoring, and identifying which approach is best suited to IBA monitoring are considered. Specific focus is paid to the applications of remote sensing in explaining species distributions, mapping land cover and land use, land cover change and a series of studies that elucidate the impact of deleterious land cover and land use change on the survival of species.

2.3.1. Remote sensing and mapping species habitats and distributions

In studies of species distributions, the establishment of direct relationships between spectral radiance values recorded from remote sensors and species distribution patterns recorded from field observations may assist in assessing distribution. Estimation of species distribution from remotely sensed data requires identification and detection of species habitat suitability. Land cover maps from remote sensing can be used as surrogates of species distribution. Globally, the distributions of individual species or sites of high species richness and diversity remain largely unknown. However, there is increasing documentation knowledge of where species are than previously, especially in the case of a very small number of better studied species, or species that occur in e.g. Europe (Buchanan *et al.,* 2009a). Earth observation data has proven to be useful in studies of species richness and diversity; both correlate with spatial and temporal variation in Normalised Difference Vegetation Index (NDVI) which is a simple numerical indicator that correlates strongly with absorbed photosynthetically active radiation and can be used to analyse remote sensing measurements, typically from a space platform, and assess whether the target being observed contains live green vegetation or not (e.g. Nøhr and Jørgensen, 1997; Hurlbert and Haskell 2003; Bonn *et al.,*2004). Diversity increases with increasing spatial variation in land cover, and more productive areas (higher NDVI) are capable of supporting more species although some suggest this relationship has an asymptote. However, this needs to be treated with caution because while NDVI may correlate with richness, it may not actually be any good at measuring conservation importance (e.g. endemism does not correlate with richness, Orme *et al.*, 2005*)*. Outputs from habitat-species association studies include systematic atlases of species distributions (e.g. Lewis and Pomeroy, 1989) but which are limited to certain parts of the world and devoted largely to birds (Gibbons *et al.*,2007).

According to Luoto *et al.* (2004), a cost-effective surrogate for deriving appropriate estimates of spatial patterns of species richness is provided by predictive modelling based on remote sensing and topographic data. Habitat types, characteristics and suitability analysis captured by land cover maps compared with species ecological niche factor analysis, habitat requirements and primary productivity is a robust approach used to infer or predict species distribution (although as with all model approaches, it is a simplification). This is possible because remote sensing data can help identify, detail the biophysical characteristics of species' habitats, predicting the distribution of species and spatial variability in species richness (Kerr and Ostrovsky, 2003). Several examples of the application of remote sensing techniques in mapping and predicting species habitats and distributions are shown in Table 4.

Table 4: Studies that have used remote sensing in species work i.e. mapping species distribution, richness

Application of remote sensing in species distributions, richness	**Methodology used**	**Key results**	**Reference**
Predicting the distribution of individual species and species assemblages	Based on species habitat suitability analysis, determined the relationship between remotely sensed habitat categorisations and species distribution patterns for plants (forest and meadow), butterflies and birds. Used remote sensing and topographic data and explained the observed distribution of threatened species and predicted the occurrence of the species in two independent test areas with different landscape structures.	Analyses of satellite and bird data have been well established for efficient ecosystem descriptions and species modelling within a large range of scales and habitats	Thibault *et al.,* 1998; Jennings, 2000; Kerr *et al.*2001; Saveraid *et al.,*2001; Conner, 2002; Hepinstall and Sader, 1991; Jenkins *et al.,* 2003a; Gottschalk *et al.,*2005; Laurent *et al.,* 2002; 2005; Luoto *et al.,*2002, 2002; Prins *et al.,*2005
Primary productivity and prediction of the distribution of species assemblages	Normalised Difference Vegetation Index (NDVI), which correlates strongly with absorbed photosynthetically active radiation is indicator for the annual biomass production	Habitat heterogeneity, as estimated by an advanced land cover classification, provides a stronger prediction of species richness in than any previously measured factor	Kerr, *et al.,* 2001, Bailey *et al.,* 2004; Seto *et al.,* 2004)
Descriptions of species-habitat associations	Habitats mapped based on spectral reflectance patterns and establishing their correspondence with the occupancy patterns of flora and fauna	Species distribution give an indication of the general biodiversity levels of the main land use categories of the study area	Nohr and Jorgensen, 1997; Jennings, 2000; Nagendra, 2001; Saveraid *et al.,* 2001; Hoersch *et al.,* 2002;
Identifying habitats important for birds	Landsat imagery was analysed to identify habitats for birds (e.g. migratory birds)	Spatial projections of species richness developed based on numbers of species from surveyed sites and relation of this information to the environmental variables sites or derived from GIS layers data produced models that yielded predictions of	Sader *et al.,* 1991; Luoto *et al.* 2004

		richness	
Assessing and predicting species richness and diversity	Using species richness data from a spatial grid system, tested the usefulness of satellite-based remote sensing and topographic data in species richness modelling	It is possible effectively link localized field investigations of biodiversity with remotely sensed information to permit extrapolations at progressively higher scales.	Nohr and Jorgensen, 1997; Nagendra and Gadgil, 1999; Luoto *et al.*, 2004; Nagendra, 2001; Oindo *et al.*, 2002
Spatial distribution of ecological communities	Using SPOT XS to map the spatial distribution of different vegetation groups classified using classification and ordination analysis using remote sensing techniques and assess the potential of SPOT XS for mapping vegetative and non-vegetative classes	Classification of vegetation groups based on species composition identified using classification and ordination techniques to some extent resemble to those groups classified using SPOT XS data with least accuracy in comparison to non-vegetation classes which were more homogenous and spectrally separable and were classified more accurately in comparison.	Malik and Husain, 2006b
Mapping the species richness and composition of tropical forests	Use habitats to map places where species are richer	Biodiversity indices of species richness and evenness derived from the remotely sensed data were strongly correlated with those derived from field survey	Foody and Cutler, 2006
Endemism	Remotely sensed environmental variables used to identify areas of bird endemism and also obtain general patterns of bird species richness	Species' ecological characteristics may influence the performance of distribution models.	Johnson *et al.*,1998

The distributions of individual species and sites of high species richness, diversity or conservation significance are increasing, well documented and remote sensing has been useful in mapping and delineating such sites worldwide. However, when dealing with the use of vegetation to map species distributions some caution is needed (Scott *et al.*, 1996). For example birds respond more to vegetation structure (e.g. canopy closure, height differentials, tree architecture) than to plant species composition while animals also vary greatly in the breadth of their habitat specificity (Wilkie and Finn, 1996). Inclusion of these structural elements in models can increase the accuracy with which distributions can be predicted (e.g. Whittingham *et al.*, 2006).

2.3.2. Remote sensing, land cover and land use mapping and classifications

Land-cover refers to the physical characteristics of earth's surface, captured in the distribution of vegetation, water, soil and other physical features of the land, including those created solely by human activities whereas land-use refers to the way in which land has been used by humans and their habitat, usually with emphasis on the functional role of land for economic activities (Kumar *et al.,* 2008). Land cover is increasingly regarded as essential information (Cihlar, 2000) and is critical for land use planning and biodiversity conservation. Analysis of remote sensed data is one of the primary methods for capturing information on land cover. Land cover and land-use mapping both by visual interpretation and digital image analysis is possible by satellite remote sensing techniques and earth observation data are widely used to identify common land cover types (Buchanan, *et al.,* 2009b*)*. Remote sensing has produced successful results for land cover mapping (e.g. Mayaux *et al.,*2004) and based on this mapping, areas of high conservation value or importance at regional scales have been identified (Brooks *et al.,*2006).

Remote sensing has been useful in generating readily available pre-processed land-cover maps such as Global Land Cover 2000 map (Mayaux *et al.,* 2004). Remotely sensed data from airborne or space-borne sensors provide a spatially extensive means to produce maps of the surface cover types and are capable of providing accurate and cost-effective information about any given area on the earth's surface at short intervals and on continuous basis (Verbyla and Richardson, 1996). Therefore these data can provide comparable, spatially explicit information over large areas (larger than can be covered by field methods), decreasing the need for extrapolation from limited samples to large spatial scales where ground-based methods may fail to capture full extents or distributions (Shuman and Ambrose, 2003). However, a combination of field surveys and remote sensing are again imperative to successfully map all vegetation distribution.

Consequently, it is possible to describe apparent habitat associations of species from land cover maps (Fuller *et al.,* 2005). Remotely sensed data has been useful in mapping biomes in detail (Bartholome and Belward, 2005). Remote sensing has also been used in the analysis of landscape patterns (Innes and Koch, 1998; Johnson *et al.*1998; Griffiths *et al.,*2000) and development of land use data that reflect human and landscape-vegetation interactions (Jakubauskas *et al.,*2002) with natural resources (Kerr and Ostrovsky, 2003).

Several examples of the application of remote sensing techniques in land cover and land use mapping and classifications are shown in Table 5.

Table 5: Examples of studies that have used remote sensing in land-cover and land-use mapping

Application in mapping of land cover	Methodology used	Key results	Reference
Tropical vegetation mapping	Using supervised and unsupervised classification methods with different band combinations to discriminate vegetation types	Unique capabilities of combined visible and SAR data for monitoring tropical vegetation, especially for forest/non-forest discrimination Even within the limit of spectral information available in the image, the digital classification can improve the result of visual interpretation.	Nezry *et al.*, 1993; Trisurat *et al.*, 2000; Curran *et al.*, 2004; Fuller *et al.*, 1998
Land-cover and land-use mapping at the rural-urban fringe	Remotely sensed data subjected to standard per-pixel classification algorithms	Integration of remote sensing and geographic information systems (GIS) has been widely applied and been recognized as a powerful and effective tool in detecting urban land use and land cover change	Treitz *et al.*, 1992; Burnett *et al.*, 2003
Classification and mapping wetland communities	Quantitative analysis (subpixel unmixing) with qualitative decision (classification), and incorporation with human knowledge Unsupervised classification or clustering, supervised classification, principal	Benefits of integrating remote sensing and ecology	Dechka *et al.*,2002; Ozesmi and Bauer, 2002

	component analysis, hybrid classification and fuzzy classification are among the most common applied techniques in wetlands classification		
Tropical forest cover mapping	Quantitative analysis of remotely sensed data, supervised and unsupervised classification	Earth observation was recognised as having considerable potential to aid an improved understanding of tropical forest biodiversity	Foody and Hill 1996; Grover, *et al.*,1999; Quegan *et al.*,2000
Land cover mapping of large areas	The classification of Landsat images based on the information in the overlapping areas of neighboring scene	Chain classification is a powerful approach for large area land cover classifications,	Cihlar, 2000; Knorn *et al.*, 2009.
Land cover map (including Global Land Cover maps)	Digital image processing and geographical information systems techniques	Map contains greater spatial detail and is more consistent than conventionally compiled maps	Phinn *et al.*,1996; Loveland *et al.*,1999; Eva *et al.*,1999; Mayaux *et al.*,1999, 2004; Hansen *et al.*,2000; FAO 2001; Stibig *et al.*, 2003; Friedl *et al.*,2002; Malik and Husain 2006a; Dash *et al.*,2007

2.3.3. Remote sensing and land cover change

Monitoring land-use and land-cover produces information about the state of the landscape for managers and conservationists (Lambin and Geist, 2006). Land cover maps are in themselves a useful tool, but are static because few items depicted on maps change as rapidly as land cover and so the ability to monitor it accurately is important (Belward *et al.,* 1999). Land use and land cover change is one of the most prevalent threats to a network of sites of high conservation importance and entails changes that could be monitored remotely (Buchanan *et al.*,2009a) and come in the form of conversion and modification. Land cover change analysis (change detection) is the process of identifying differences in the state of an object or phenomenon by observing it at different times (Singh, 1989; Aplin, 2006). The potential of earth observation in biodiversity conservation and monitoring has been recognised (Turner *et al.*, 2003; Pettorelli *et al.*, 2005) and satellite image analysis has been used to detect natural and human-induced changes and threats within and across landscapes (Kerr and Ostrovsky, 2003; Buchanan *et al.*, 2009a). Initial land-cover change analyses were in tropical forest areas (e.g. in the Amazon forest, Tucker *et al.*1984, Skole and Tucker 1993) setting the stage for tracking changes in land cover through satellite images, which allow conservation problems to be identified and solutions developed. However, besides tropical forests, land cover change analyses for all land cover types (rangelands, open forests, urban areas, wetlands) are now becoming understood (Lambin and Geist, 2006).

Applications of remote sensing in change detection analyses are not only widespread but also becoming popular because of the availability of a long time series of remotely sensed imagery dating from 1970s to date. Remote sensing provides an excellent source of data for change detection analysis because of the continuous and stable nature of space borne image acquisition (Lu *et al.*, 2003; Symeonakis and Drake, 2004; Coppin *et al.*, 2004). For example, change detection uses the pixel-to-pixel analysis whereby a series of images acquired over time are subjected to trend analysis (Collins and Woodcock, 1996; Franklin, *et al.*, 2000).

The efficacy of remote sensing in land cover and land use change is based on the fact that remote sensing is useful in the analysis of landscape patterns (Innes and Koch, 1998; Johnson *et al.*, 1998; Griffiths *et al.*, 2000) and development of land use data that reflect human and landscape-vegetation interactions (Jakubauskas *et al.*, 2002) with natural resources (Kerr and Ostrovsky, 2003). Studies conducted using remote sensing provided qualitative assessments of habitat (Mutanga and Skidmore, 2004), deforestation rates and its impacts on species i.e. endemic birds and range restricted birds (Buchanan, *et al.*, 2008)

Remotely sensed data methods as applied to assess spatio-temporal land cover change, forest fragmentation or disturbance at various habitat types have emerged as suitable means for gaining information on the current state of habitats as well as the disturbance history. The technique has been used in change detection analysis of various ecosystems, i.e. forest cover (Hame *et al.*, 1998). Many land cover change assessments have predominantly focussed on forests (e.g. extensive documentation of global deforestation, Wade *et al.*, 2003). This is because patterns of forest fragmentation are relatively easy to pick out due to the distinct differences in the reflectance of forested and cleared land (de Sherbinin, 2005). Focus on deforestation is perhaps attributed to the fact that changes in forest cover over time are important because of the role forests play in the global carbon cycle, in global climatic trends, and in providing species habitat (Woodwell *et al.*, 1984). In forest ecosystem studies, remote sensing has also been used to monitor land cover change ranging from deforestation (Tucker *et al.*, 1984, Malingreau *et al.*, 1988; Stoms and Estes, 1993; Turner *et al.*, 1995; Sayre *et al.*, 2000) to fragmentation and edge effects (Skole and Tucker, 1993; Achard *et al.*, 2002). Remote sensing has not been confined to terrestrial ecosystems dynamics only. Instead, it has also been used in marine and aquatic studies ranging from analysing ecosystem decline along a river basin (Twumasi and Merem, 2006), to assessing change within a coastal marine ecosystem (Chauvaud *et al.,* 1998; Weng, 2002; Twumasi and Merem, 2007).

Causes of land-cover change at various scales are becoming well documented and remote sensing is also a useful technique for identifying drivers of ecosystem change and biodiversity loss. However, land cover change may not always be negative but could also be positive manifestation of focused site interventions, i.e. in habitat restoration efforts. Evaluating the impact of conservation management interventions is crucial to the process of maximising conservation outcomes. The use of remote sensing in forest and wetland restoration is well documented. Most of these have focused to classifying patterns of forest and wetland succession or restoration (Walker *et al.,* 1986; Hall *et al.,* 1987).

However, applications of land cover change using remote sensing are replete with forest studies and therefore the need to diversify the use of this technique in monitoring other habitat types so as to have a complete representation of land cover and land use trends in a whole suite of habitats. This may explain why tropical deforestation has been stated as one of the primary causes of global environmental change (Geist and Lambin, 2002). For example, accurate, efficient, and repeatable mapping of changes in wetlands is critical for monitoring natural, anthropogenic and climatic impacts on these critical systems. There are many studies focusing on grassland ecosystems, but many focus on change as a result of ecological interactions (e.g. Milton and Dean, 2001). Despite the savannah ecosystem covering 20% of the earth surface and 40% of Africa (Scholes and Walker, 1993) and their critical roles for biodiversity conservation, only a few of these studies have been useful for conservation (e.g. Jepson, 2005) or developed methods that are transferable. This is against the backdrop of many threats facing grasslands in many locations in Africa (e.g. MacDonald *et al.,* 1993, Reyers *et al.,* 2001, Ndang'ang'a *et al.*, 2002).

Several global land cover data sets derived from satellite observations have become available to the scientific community and even though they offer valuable information on the current state of the Earth's land surface, considerable disagreements among them and classification legends not primarily suited for specific applications such as carbon cycle model parameterisations pose significant challenges and uncertainties in the use of such data sets (Jung, *et al.,* 2006). The temporal, spatial and spectral resolutions of the remote sensing data will affect the changes that can be detected. As such, even though global maps with coarse spatial resolution provide a reasonable snapshot of the general extent of forest cover, large scale products such as Global Land Cover 2000 (Bartholomė and Belward, 2005) and Globcover cannot be compared and are not suitable for global or national forest operational monitoring products such as Global Land Cover 2000 (Bartholomė and Belward, 2005) and Globcover cannot be compared and are not suitable for global or national forest operational monitoring because of insufficient accuracy and the lack of detailed forest themes (Strand *et al.,* 2007). For example, the GLC2000 product includes eight forest classes. Differences in resolution (spatial, temporal and spectral), ambiguity in signatures of land classes in the classification legends and accuracy are among the issues that need to be addressed before they can be used for land cover change analyses (Jung *et al.,* 2006*)*.Various change detection approaches are being applied but with varying degrees of shortcomings that limit the accuracy of classification based change detection. As described by Castelli *et al.*, (1999), these shortcomings are presented in the box below.

The process of classification based change detection is also compounded by the fact that commonly used change detection algorithms fail to take account of spatial information and hence it is not possible to pick up differences in reflectance of arable versus herbaceous unlike with the visual interpretation methodology.

1. Limited spectral separation of classes
2. The separation of the classification, and change detection steps: Change detection step uses the processed information from classification, not the original data,
3. The "statistical independence" assumption: Pixel-wised classifications involve the values individually without considering the neighbouring pixels,
4. The construction of training sets: Collection of ground truth is expensive, time-consuming, and sometimes it is impossible to assign the class label for spectral clusters. A less expensive alternative is obtaining training sets from photo interpretation that limits the resulting accuracy to the skills of human the expert, who constructs the training sets,
5. The classification taxonomies are often based on land-use that cannot be identified using remotely sensed data in some cases, rather than land cover type (e.g., discrimination between reservoir and lake). Also, different types of land cover are often aggregated under a single class. The intrinsic limitation of classifiers-Comparison of multi-resolution/multi-temporal image data in a change detection procedure requires more research that would involve new approaches of multi-scale analysis.

A review summary of the various applications of remote sensing in land cover and land use changes is presented in Table 6.

Table 6: Examples of applications of remote sensing in land cover and land use change studies

Application in mapping of land cover change	**Methodology used**	**Main results**	**Reference**
Forest cover change, deforestation	Land cover classification and digital change detection techniques and visual delineation and interpretation of multi-temporal and multi-resolution remotely sensed data	Deforestation and forest degradation increased substantially	Tucker *et al.*,1984; Skole and Tucker 1993; Souza and Barreto, 2000; Steininger *et al.*,2001; Zhang *et al.*,2001; Achard *et al.*,2002; Asner *et al.*,2002; Batistella *et al.*2003; Kinnaird *et al.*,2003; Linkie *et al.*,2004; Curran *et al.*,2004; Lung and Schaab, 2004; 2006; Asner *et al.*, 2005; Buchanan *et al.*,2008.
Forest fragmentation	change detection algorithms,	Fragmentation of habitats due to forest loss and degradation	Skole and Tucker, 1993; Carvalho and Vasconcelos, 1999; Petit and Lambin, 2001; Achard *et al.*, 2002.
Wetland change	Supervised and un supervised classification and change detection algorithms, visual interpretation of remote sensed data	Wetland vegetation cover declined as a result of anthropogenic activities	Ramsey and Jensen, 1996; Munyati; 2000; Houhoulis, and Michener, 2000; Sajeev and Subramanian, 2003; Baker *et al.*, 2006, 2007; Ernani and Gabriels, 2006; Owino & Ryan, 2007.
Mapping and monitoring vegetation and land cover change	modelling procedure, automated classification, and change detection, multi-temporal principal components analysis	Varying patterns of vegetation change	Aspinall and Veitch, 1993; Fuller *et al.*, 1994; Steven and Jaggard, 1995; Zheng *et al.*, 1997.
Mapping fire induced land cover change	transformation techniques followed supervised classification, digital change detection techniques	Seasons of high fire frequency identified, susceptibility of vegetation types to fires noted	Li *et al.*,1997; Liew *et al.*,1998; Patterson and Yool, 1998; Pereira *et al.*, 1999; Eva and Lambin, 2000; Kiran *et al.*, 2006,
Monitoring vegetation change in protected areas	Analysis of multi-temporal NDVI composites metadata analysis of locational information	Variation in phenologies, changes inside protected areas showed lower rates of clearing/land cover loss	Batista *et al.*, 1997, Vasconcelos *et al.*,2002; Nagendra, 2008

Assessing desertification as a driver of land cover change	NDVI, Moving Standard Deviation Index (MSDI) and land surface albedo were selected as assessment indicators of desertification to represent land surface conditions from vegetation biomass, landscape pattern and micrometeorology	Expanding and decreasing trends in desertification respectively. Locations desertified and at risk of desertification identified	Ali and Bayoumi, 2004; Xu *et al.*, 2009
Invasion of alien species	Transformation, correction, and normalization of Hyperion reflectance image data into composition images with a subpixel extraction model and digital image analyses	Spatial distribution of invasive species recorded and invasive and weed species distinguished	Ramsey *et al.*, 2005; Anderson *et al.*, 1993.

2.3.4. Land cover change and impact on species

Changes recorded from remote sensing images have been widely applied and demonstrated to be capable of monitoring land cover and by association of animal species across regions and hence useful in the field of biodiversity conservation (de Sherbinin, 2005). Studies demonstrate the link between land cover changes and decline in species richness. For example, there are studies to illustrate impacts of land cover change through deforestation and fragmentation and its impacts on endemic birds and range restricted birds, breeding success of species (Kurki *et al.*,2000*)*, and endangered species e.g. large mammals (Kinnaird *et al.*,2003). Many studies have been undertaken to demonstrate how deleterious land cover changes affect species and biodiversity at large. A summary of the review of some of the studies that have used remote sensing to assess land cover and correlate the changes to species diversity and richness are presented in Table 7.

Table 7: Examples of land cover change studies and impact on species

Impact of land-cover change on species	Methodology used	Key results	References
Fragmentation of tropical forests and decline in species richness	Species-area relationship, linear models and a formal meta-analysis to predict rates of species loss in a fragmented forests	Species loss was higher in smaller fragments	Lovejoy *et al.*, 1986; Bierregaard *et al.*, 1992; Fahrig, 1997; Hargis *et al.*,1999; Chittibabu and Parthasarathy, 2000; Kurki *et al.*,2000; Fahrig, 2002; Jha, *et al.*,2005.
Forest fragmentation and nesting success of migratory birds	percent forest cover of patches compared with incidences of nest predation and parasitism	Nest predation and parasitism by increased with forest fragmentation and reproductive rates were low enough for some species in the most fragmented landscapes	Robinson and Wilcove, 1994; Robinson *et al.*, 1995.
Forest fragmentation and litter dwelling species	Responses of ants nesting in twigs in the litter layer to habitat changes associated with forest fragmentation assessed	species richness of ants and nest densities was greater in continuous forest than in fragments hence edge and isolation effects both play a role in structuring litter-dwelling ant communities	Carvalho and Vasconcelos, 1999.
Ecological impact of land cover change	Satellite image interpretation and field verification in order to identify causes of changes. principal component analysis (PCA) was used to quantitatively study driving forces of forest land use change	Net decline in ecosystem services of forest Increase in species richness in forest fragments compared to their species richness before isolation Edge effects in modifying forest dynamics	Gascon and Lovejoy, 1998; Jules *et al.*, 1999.
Habitat loss and species loss and extinction	estimating the relative importance of habitat loss and habitat spatial pattern (fragmentation) on	effects of habitat loss far outweigh the effects of habitat fragmentation Habitat loss threatens forest-obligate birds	Fahrig, 1997; Harris and Pimm, 2004.

	population extinction, using a simple, spatially explicit simulation model		
Habitat change and species response	evaluated changes in the distribution sizes and abundances of butterflies species where changes in climate and habitat are opposing forces	dual forces of habitat modification and climate change are likely to cause specialists to decline, leaving biological communities with reduced numbers of species	Warren *et al*., 2001

2.3.5. Remote sensing: The caveats to note

There is no doubt that remote sensing is valuable for ecological and biological applications and will play an increasingly important role in the future (Kerr and Ostrovsky, 2003) and that with increasing spatial, spectral, temporal, and radiometric resolutions, it will become more widely used in ecology and biodiversity conservation. Remote sensing offers the opportunity for large area characterizations of biodiversity in a systematic, repeatable, and spatially exhaustive manner (Duro *et al.,* 2007).The analysis relies upon changes in land cover altering the reflectance characteristics measured by the satellite sensor. Consequently, it will not be able to detect changes that do not alter these patterns (e.g. changes under canopies). Similarly, it will not be able to measure all threats to sites (e.g. hunting). Other constraints include technical limits on feature discrimination, the requirements of high levels of technical expertise, costs (although cheaper than field-based assessments) and the need for information to calibrate and verify remote sensing results, which can be limiting (Turner *et al.,* 2003*).*

Many digital change detection algorithms have been developed to reveal changes since the launch of ERTS-1 in 1972 (Fuller, 2006). The complexity of change detection procedure depends on the characteristics of data sets. The quality of the outputs from change detection analyses depend on various factors. For example, the selection of a single sensor series, low cloud cover and matching dates of two image datasets can restrict this intricacy when differences between spatial resolution and spectral band pass of two image dates acquired with two sensors complicates direct comparison of data to detect changes.

Uhlir (1995) warned that remote sensing technology and the information derived from it cannot be considered as a technological panacea. For example, the accuracy of the data (e.g. compromised during information gathering or processing phases, (Uhlir, 1995), results and outputs from remote sensing studies also depend on the user's knowledge of the land cover history and spectral characteristics of the area. Cloud and haze can severely preclude image interpretation in the tropics where cloud cover is common and can distort feature spectral signatures resulting in greater error or expensive and complex processing. Also, the timing of image acquisition relative to field surveys is very important because landscape features are not static but change in response to weather patterns as well as unpredictable human and natural disturbances (Wilkie and Finnie, 1996). Additionally, for most sensors, remote sensing can monitor only features that can be viewed from above and therefore the characteristics of the understorey must be inferred rather than directly observed (Strand *et al.*, 2007).

The availability of historical data (analogue or digital) is necessary while, according to Aplin (2006), temporal resolution is significant where imagery is used to monitor changing environmental conditions. For multi-temporal analysis, images should at least be of the same area at two or three points in time (temporal resolution). Images should be cloud free, spatial resolution should be fine enough for accurate mapping and classification and last but not least, they should not only be affordable but preferably, the focal area should be covered on a single image. Finally, the cost of software, and the analytical skills needed to use the images safely is a limitation where funding and capacity are limited. If these technologies are to be used widely, simple methods of analysis are needed (Buchanan *et al.,* 2009b). Nevertheless, these few limitations notwithstanding, the capabilities and application of remote sensing in biodiversity conservation has steadily grown in the past and is poised to increase dramatically as a result of improvements in the software tools, computing power, capacity in skilled personnel, increasing quality and quantity and the ever reducing costs in satellite imagery. There exist enormous strengths, opportunities to be exploited, weaknesses and threats (de Sherbinin and Giri, 2001) to be addressed and it is encouraging to note that most of the weaknesses and threats are being addressed through rapidly reducing costs of data, advances in improved accessibility and capacity to manipulate data as well as increasing accuracy and data continuity.

CHAPTER THREE: METHODOLOGY

3.1. Study Area

Site selection

A sample of 80 IBAs was originally selected for the study. However, due to incompleteness of various satellite images, only 71 of the 1,230 IBAs in Africa were used (Annex 2). These were all within East Africa, with 4 in Burundi, 8 in Ethiopia, 25 in Kenya, 2 in Rwanda, 18 in Tanzania and 15 in Uganda. According to Le Houerou and Hoste (1977) the vegetation of East Africa is highly heterogeneous, with great variability in structure and productivity, the latter greatly influenced by the spatial and seasonal distribution of precipitation and temperature. A balance across land cover types was required to ensure equal representation of the major habitats within IBAs in the study region. The dominant habitat within each IBA according to GLC2000 (Mayaux *et al.*, 2004) and BirdLife International (2008) was used to identify the dominant land cover type on IBAs, and a sample of each used (Figure 2). Thirty one were dominated by forests, 18 wetlands, 13 grassland (both savannah and highland grassland), 1 shrub lands, and 8 transition/natural-agricultural mosaic sites. Accurate digital boundaries were available for all sites and a mixture of both protected and non-protected areas was considered. Based on dominant land cover type as described by BirdLife International (2008a), six broad land cover/habitat types (artificial landscape, forest, grassland, natural-agricultural mosaic, shrub land and wetlands) for each IBA were determined.

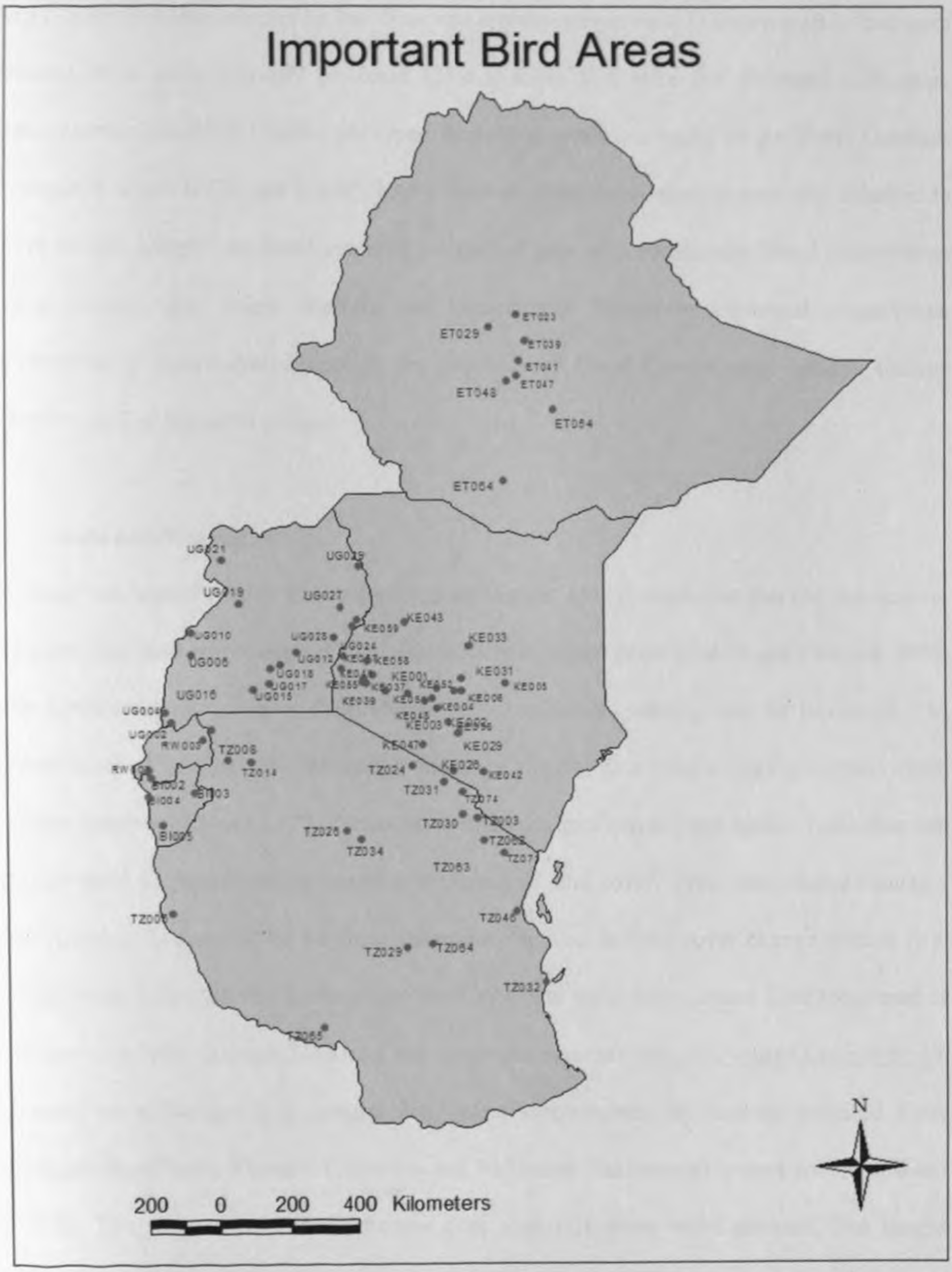

Figure 2: Map showing all the focal sites in Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda used in the study.

The 71 IBAs were also selected so that there was a reasonably even split between IBAs that were Protected Areas (41), partially protected (2) and those that were not protected (28) again balanced where possible by land cover types. Protection status was based on the World Database of Protected Areas (IUCN and UNEP, 2008). The sites that were selected were also balanced to ensure that the sample contained a similar number of sites with community based conservation efforts, namely sites where BirdLife has implemented long-term site-based conservation interventions as demonstrated through the presence of Local Conservation Groups (hereby referred to as Site Support Groups).

3.2. Data and Sources

The ready and free availability of multiple epoch images, spatial resolution that can characterise land cover and land use change and its affordability in recent years (Cohen and Goward, 2004) made Landsat satellite imagery the preferred choice of remote sensing data for this study. The Landsat series of satellites has the longest stream of satellite data from a single platform, which has been operational since 1972. It also has a relatively medium to high spatial resolution and therefore good for investigating temporal dynamics in land cover. This and other advantages makes Landsat imagery to be the most commonly applied in land cover change studies (e.g. tropical forest, Fuller 2006). Landsat data from 1990 and those from around 2000 were used as these were available through NASAs Land cover programme (http://www.usgs.gov/pubprod/). The more recent images from around 2007 were supplemented by recently acquired Aster (Advanced Space borne Thermal Emissions and Reflection Radiometer) images for the mid and late 2000. This allowed land cover changes over almost 20 years to be assessed. The images from around 1990s were captured by Landsat-TM (Thematic Mapper) with a spatial resolution of

28m, while post 1999 images generally were collected by Landsat-ETM+ (Enhanced Thematic Mapper Plus), again with a spatial resolution of 28m. Scenes from these epochs are available free of charge for download from the University of Maryland's Global Land Cover Facility (GLCF; http://www.glcf.umd.edu/index.shtml). Scenes for the recently acquired Aster for the mid 2000s were purchased to complement Landsat ETM+ for mid and late 2000s. These were used at 30m resolution. Most of these images are already orthorectified and available as Geocover datasets. On acquisition, the data had already been pre-processed through georeferencing and radiometric corrections.

3.2.1. Human population data

Human population densities (per square km) covering 1995 and 2000 were reprojected to a 1 km resolution from the Gridded Population of the World (GPW) website (CIESIN and CIAT. 2005). The Gridded Population of the World (GPW) consists of estimates of human population for the years 1990, 1995, 2000 and 2005 into a global grid of quadrilateral latitude-longitude cells at a resolution of 2.5 arc minutes. These population density data are far more precise than in most other such global analyses, which often use countrywide averages as the data (Harcourt and Parks, 2003). The data are presented in the site as a grid with a cell size of 0.083x0.083 degrees (or 50x50 minutes, or approximately 9 kmx9 km at the equator). The 'smoothed' version of the data in the site was used because it removes abrupt transitions in density at political boundaries, and thus provides a more realistic distribution of densities.

3.2.2. Protected Areas data

The Protected Area status for each site in this study was derived from the World Database on Protected Areas (IUCN and UNEP, 2008). Areas that were designated under categories I-VI based on primary management objectives (IUCN, 1994) were classified as Protected Areas. The categories used include Strict Nature Reserves (including Forest Reserves), Wilderness Area (including Game Reserves) and National Parks. Where applicable, the years when they were designated and duration since designation were also recorded to document the protection status history.

3.2.3. Species data

Data on species population trends and changes in the Red List status of species occurring inside IBAs were acquired from the BirdLife International database and from a review of existing reports and publications for comparison to land cover change as assessed by this study. These data were available for only a sub set of five sites as such data does not actually exist for many sites.

3.3. Overview of Conservation Intervention

Information on past interventions was used to assess whether land cover change is less frequent following long-term site-based conservation initiatives. Based on consultations within BirdLife International and Partners in the respective countries as well as a thorough review of existing reports and project and programme documents, the sites where BirdLife and its Partners have been working in the last decade were identified. These include sites where there have been long-term conservation intervention activities such as actual site-based conservation initiatives, long-term engagement with local communities (herein referred to as Site Support Groups) through community-based natural resource management and livelihood improvement initiatives. The number of years for which this intervention or engagement has been undertaken at each respective site was determined. The presence of an SSG is an indicator of BirdLife interventions at the site and as such it was assumed that SSGs through their conservation work improve the condition of the site and prevent further loss, decline or deleterious land cover change.

3.4. Image Interpretation and Analysis

A systematic sampling approach (e.g. Achard *et al.,* 2002, Brink and Eva 2009) was used to assess land cover change. Land cover, and land cover changes, were determined by visual interpretation of land cover within sample boxes (300 m by 300 m) overlayed onto satellite images from all epochs, distributed at equal intervals across IBAs, and across a 20 km buffer around the IBAs. Within IBAs, the boxes were separated by 1.5 km in IBAs larger than 50km^2 (62 IBAs), and 0.5 km in the (10) IBAs smaller than this (to ensure sufficient numbers of points for analysis). Boxes were separated by 3 km in the 20 km buffer around IBAs. Visual Interpretation of satellite images was done using a tool developed by EC-JRC (D. Simonetti, unpublished) that was designed to aid quick visual interpretation of high resolution multi-temporal satellite data as shown in Figures 3, 4, 5 and 6.

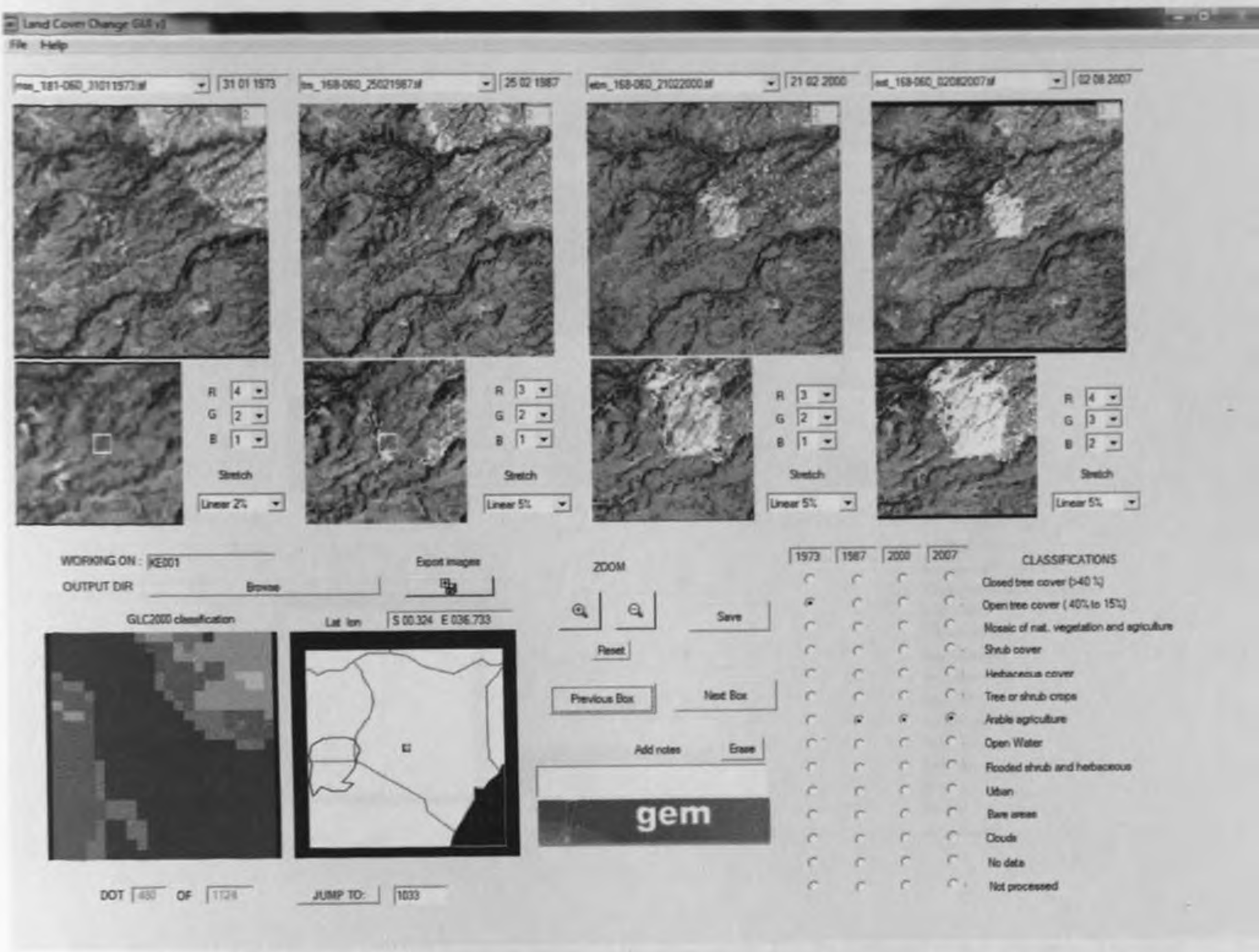

Figure 3: Example screen shot of GUI used to assess forest land cover change at the Aberdare National Park, IBA. The four boxes cover, from right to left, the different time periods used in the study

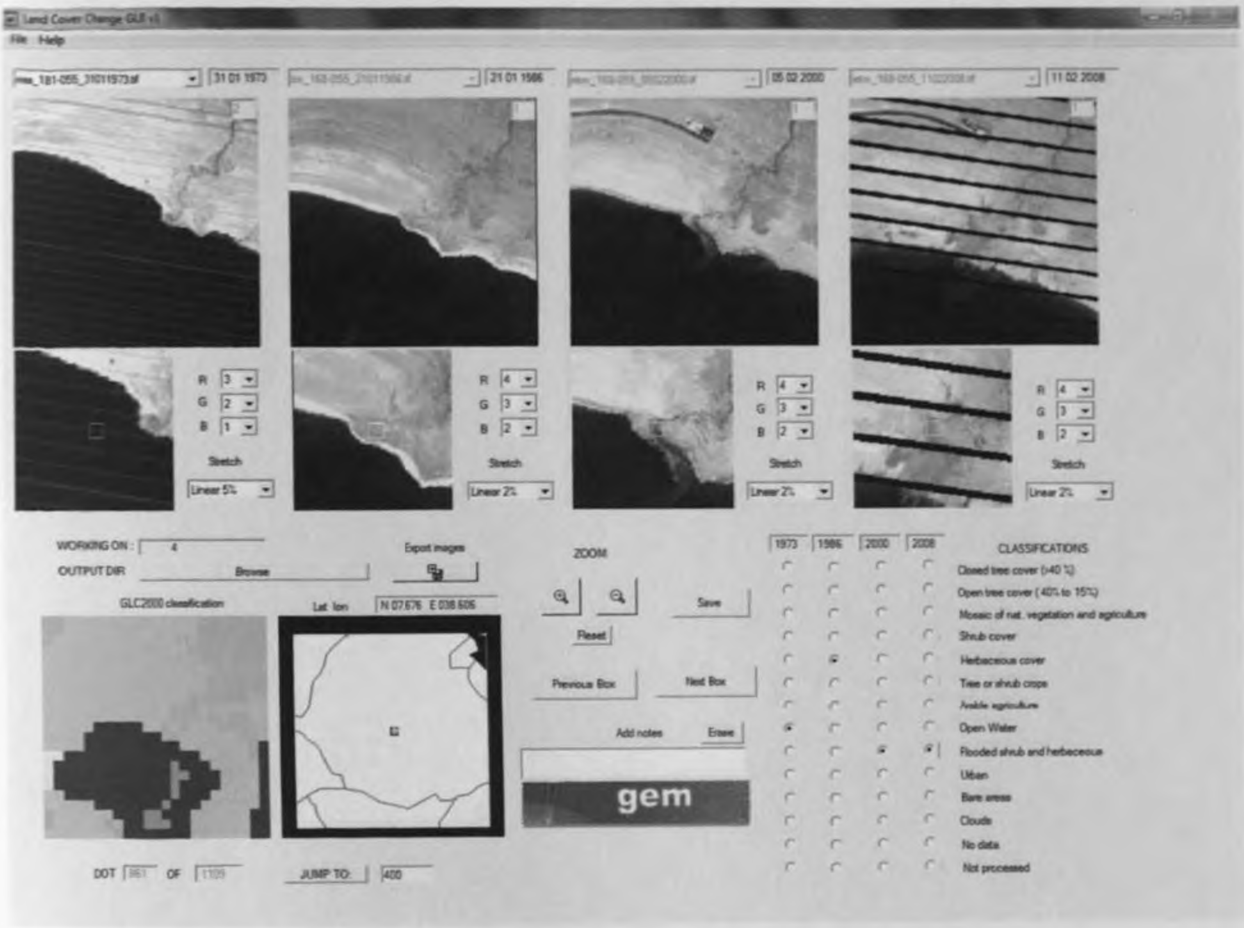

Figure 4: Example screen shot of GUI used to assess Abijatta-Shalla Lakes wetland cover change. The four boxes cover, from right to left, the different time periods used in the study

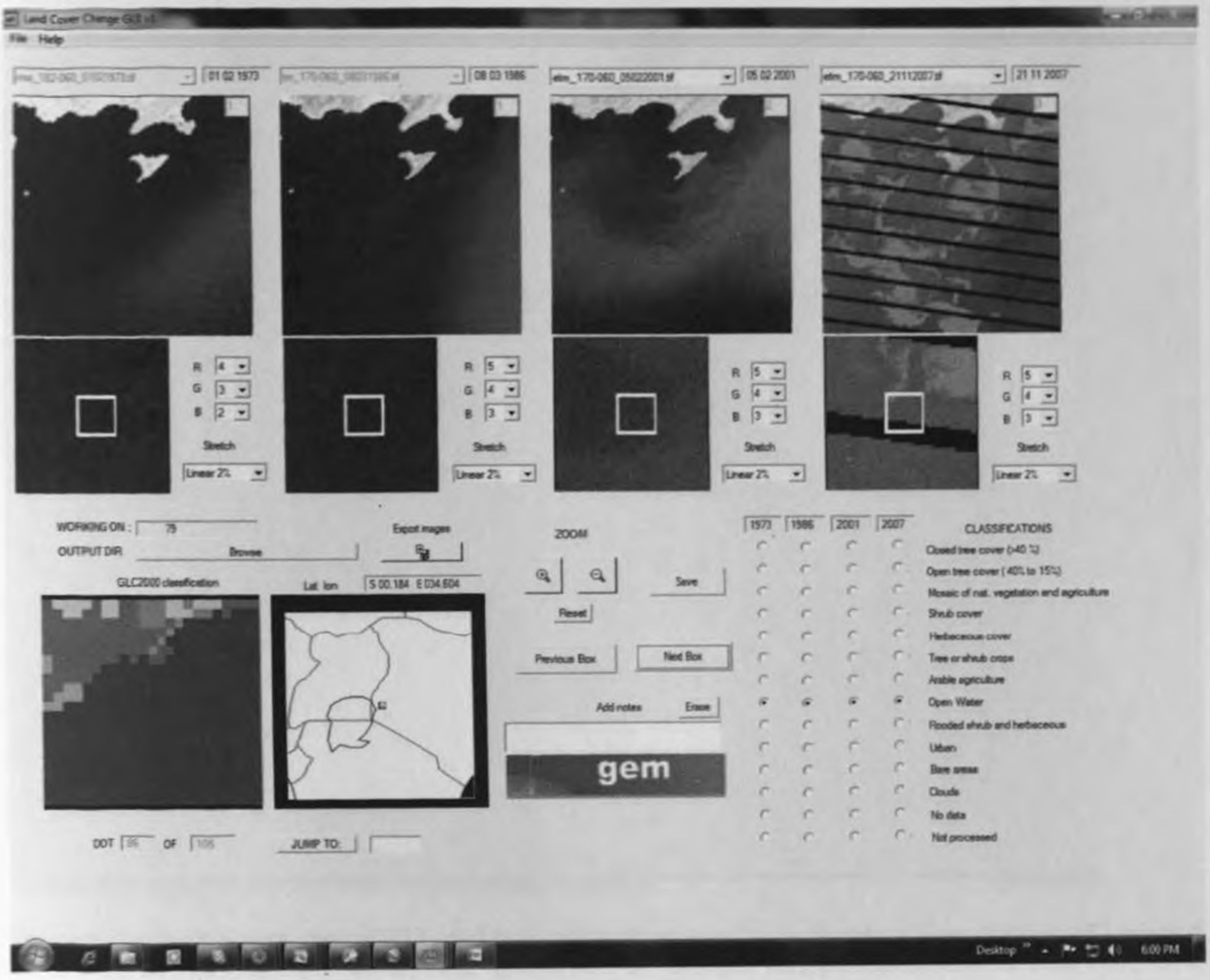

Figure 5: Example of the invasive water hyacinth spread in Lake Victoria in box four

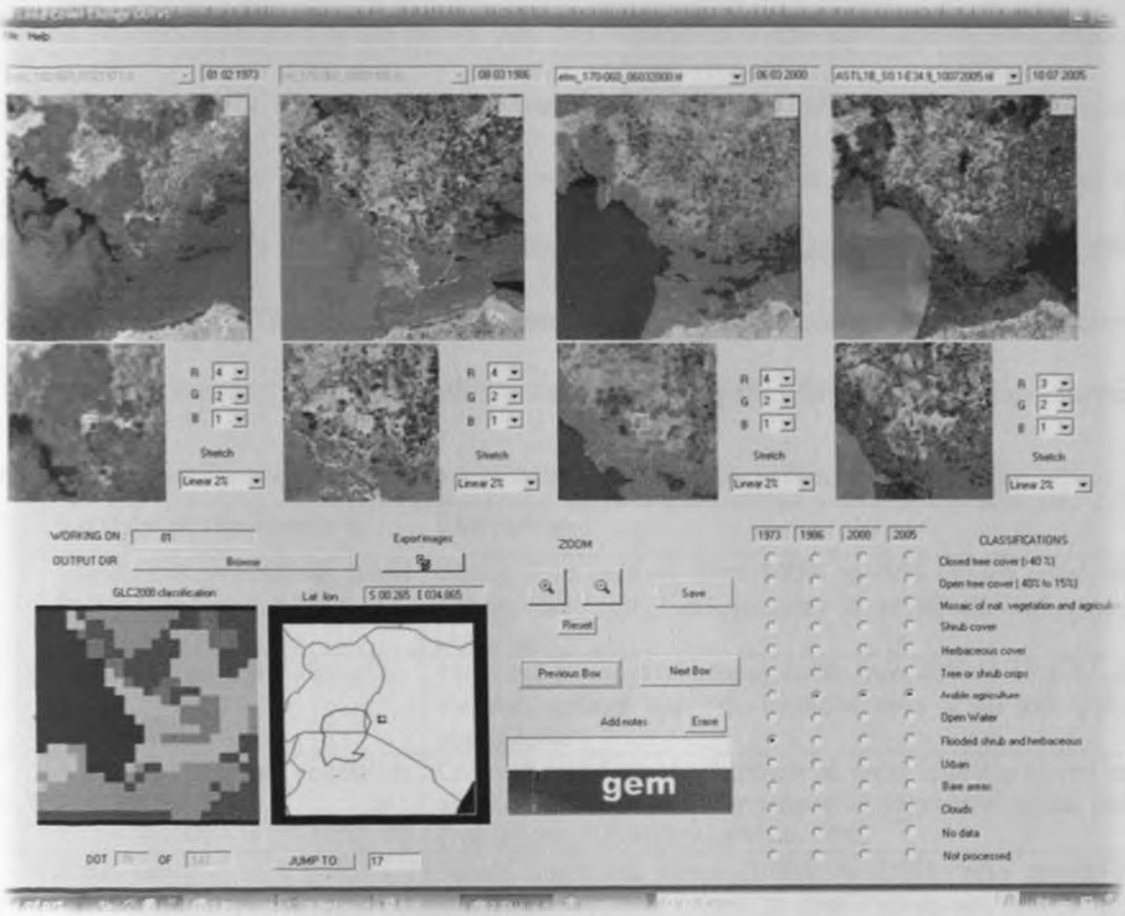

Figure 6: Example screen shot of GUI used to assess wetland land cover change at Kusa Swamp IBA around Lake Victoria. The four boxes cover, from right to left, the different time periods used in the study and show a decline in flooded vegetation

It has four viewing windows displayed; each with a zoom capability window and a total of ten generalised land cover classes chosen for each year of available imagery. Interpretation for each box was based on the predominant land cover class within the 300m by 300m box. The dominant land cover class was identified as one of closed or dense forest (>40% cover), open or degraded forest (<40%), natural vegetation-agricultural mosaic (15%-65% mixture), shrub lands, grasslands/herbaceous, agriculture-plantation (tree/shrub crops or managed non-arable agricultural vegetation such as tea, coffee, trees), agriculture-arable (small scale subsistence agriculture to large scale fields), open water (standing or flowing with no observable vegetation), flooded vegetation (clearly more humid than the vegetation in the surrounding area, associated with a watercourse or a ready supply of water) , bare ground (no sign of vegetation), cloud, urban, unknown or no data. These classes were defined based on the hierarchical land cover classes from the FAO Land Cover Classification System (Di Gregorio and Jansen, 2000) as presented in Table 8. This system is designed to produce objective assessments of land cover on a hierarchical classification system and is increasingly being used for remote sensing analysis.

Table 8. The Various land cover categories used during the visual interpretation of satellite data

No.	Land cover classification	Description
1	Closed tree cover/dense forest (>40%)	Closed Forest Cover, from 100% to 40% canopy cover. Height greater than 3m and includes trees of all leaf types and phenologics
2	Open tree cover (40-15%)	Open tree cover, with canopy cover from 40% to 15%. Height of the trees greater than 3m. Includes trees of all leaf types and phenologies
3	Mosaic of Natural vegetation and Agriculture	Cultivated and managed terrestrial areas spatially mixed natural and semi-natural primarily terrestrial vegetation. Class includes proportions of both from 65% to 15%
4	Shrub cover	Shrub dominated cover (100% to 15%), with possibility of additional layers of closed to open herbaceous (100% to 15%) and/or open tree cover (<15%) and height <3m
5	Herbaceous cover	Herbaceous dominated cover (100% to 40%) with possibility of additional layers of open shrub (<15%)
6	Tree or Shrub crops	Plantations or orchards of tree or shrub crops
7	Arable agriculture	Arable agriculture including small, medium and large scale field sizes. Can also include fallow fields
8	Open water	Standing and flowing water. The presence of water being any combination of permanent, seasonal or temporary
9	Flooded shrub and herbaceous	Flooded vegetation which can include a thematic mix of shrub, herbaceous and moss or lichen cover. The presence of water being any combination of permanent, seasonal or temporary
10	Urban	Areas extensively modified by humans, but not covered by agriculture. Can include built up and non-built up areas, urban vegetated areas, and extraction sites
11	Bare areas	Percentage vegetation cover is less than 4% and includes bare soil, bare rock, stoney and sandy cover

To aid in the image interpretation process, Google Earth was also used in supplementing the validation process because of its capability to provide a very high spatial resolution snapshot of an area. Post-processing image interpretation involved textual outputs in summary sheets and tables on which statistical analyses were based.

3.5. Ground-truthing

Although aerial imagery provides a valuable source of information, field observations are important for accuracy assessment. Field observations are required for effective and accurate image interpretation as it makes it possible to ascertain the relationships between the landscape and its appearance on the image. Accuracy is considered to be the degree of closeness of results to the values accepted as true. Accuracy assessments determine the quality of the information derived from remotely sensed data (Congalton and Green, 1999). Accuracy assessment requires the collection of a set of reference data consisting of points of known identity that are dispersed throughout the area to be studied and allocated across the classes used for the map (Aronoff, 2005).

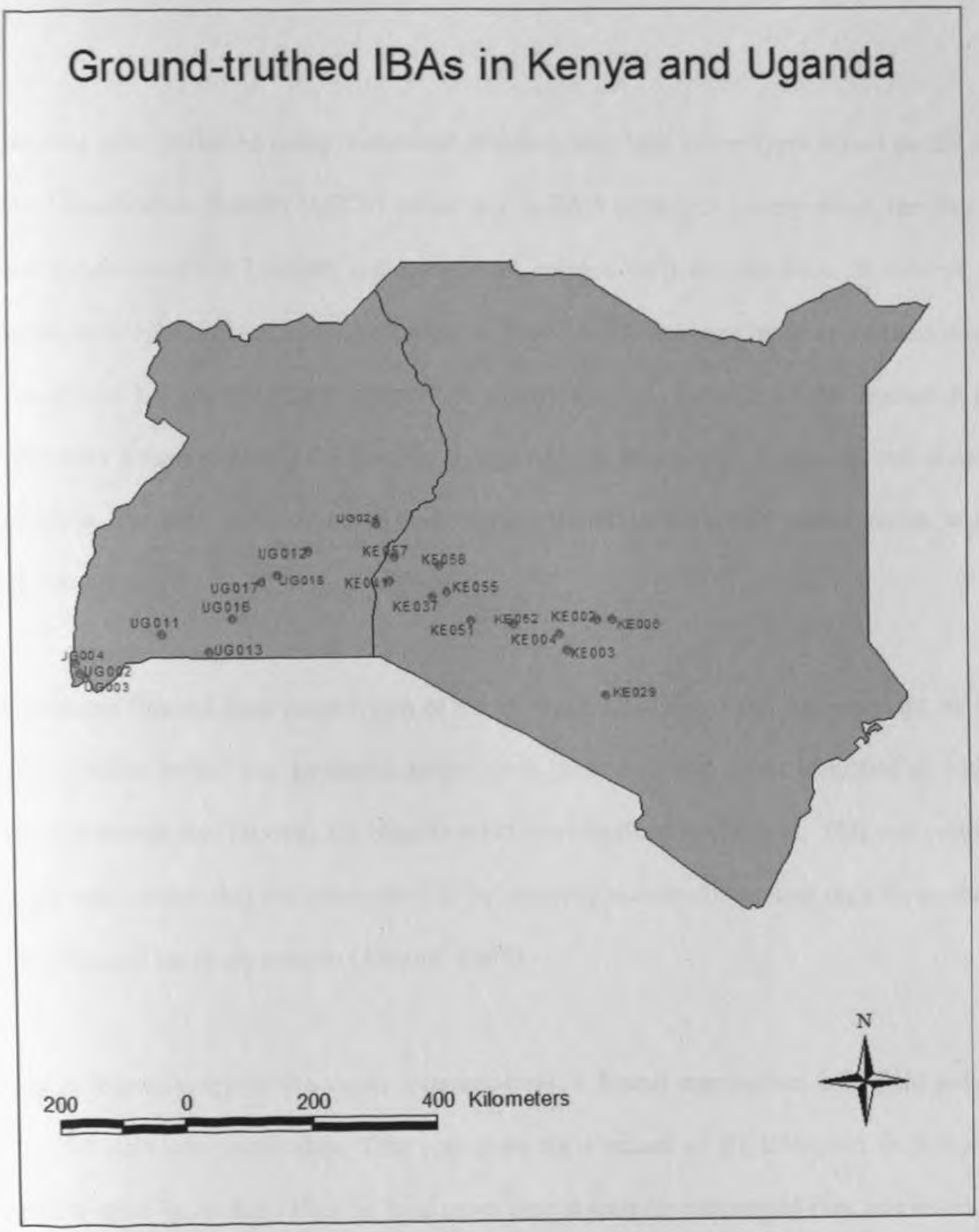

Figure 7: Map showing the sites where ground-truthing was conducted in Kenya and Uganda to aid the accuracy assessment process (site UG002 was not included in the land cover change analysis).

These data were collected using a standard protocol with land cover types based on the Land Cover Classification System (LCCS) as set out in FAO (2005). At every point, the data was spatially referenced (in Latitude and Longitude) using a GPS set. At least 30 surveys were undertaken in the main land cover categories on the IBA. Efforts were made as much as possible to cover most habitat types representative of a particular site. Samples of the vegetation types present were dispersed across the IBA and separated by at least 200m. Pragmatic considerations were made regarding the location of these survey locations, especially where access to some areas was impossible.

The locations (but not land cover types) of the ground-truthed data were imported into ArcMap. A 100 m radius buffer was generated around each point, and land cover identified as with the land cover change tool (above), but blind to what was identified in the field. This was consistent with the requirement that the reference data for accuracy assessment be used only for evaluation and not as input for interpretation (Aronoff, 2005).

To assess the accuracy of the visual interpretations, a formal comparison with field collected land cover data was undertaken. This was done for a subset of 21 IBA sites in Kenya and Uganda (Figure 7). At these sites the land cover type at spatially referenced sites was noted. This independent dataset for land cover/land use types was collected from 21 sites. Plates 1-8 present snapshots taken in the field during the ground-truthing exercise in Kenya and Uganda. Different measures of accuracy used, including overall accuracy, the confusion matrix, Producers and User's accuracy; error of commission/omission, and the kappa coefficient were examined in the error analysis. Overall accuracy was calculated by dividing the number of validation pixels that were classified correctly by the total number of validation pixels for all classes. The error matrix (also referred to as confusion matrix or contingency table) was used to illustrate class agreement and error in greater detail by illustrating the relationship between the independent validation sites (of the known class) and the percentage of those pixels actually classified into the various classes by the maximum likelihood classifier (Jensen, 2000; Lillesand and Kiefer, 2000). Percentages of pixels classified correctly are shown on the diagonal of the confusion matrix, while errors of commission (incorrect inclusion into class-row entries) and omission (incorrect exclusion from class column entries) can be seen off the diagonal (Lillesand and Kiefer, 2000). The producer's accuracy which is a measure of omission error was computed by dividing the number of correctly classified pixels in each category by the total number of pixels in the corresponding column. The user's accuracy which is a measure of commission error was computed by dividing the number of correctly classified pixels in each category by the total number pixels that were classified in that category (corresponding row total). The same methodology has been used by Malik and Housain (2006a).

Plate 1: Field Assistants collecting data in disturbed forest, Kenya

Plate 2: Field Assistant collecting ground-truthing data around Doho Rice Scheme IBA site in Uganda

Plate 3: Large scale sugar cane farming is one of the common land uses around Mabira Forest IBA, Uganda

Plate 4: Field Assistants deep inside the secondary forest, Mabira Forest, Uganda

Plate 5: Field Assistants collecting data at the Kinangop Grassland IBA, Kenya

Plate 6: Data collection involved a lot of trekking

Plate 5: Field Assistants collecting data at the Kinangop Grassland IBA, Kenya

Plate 6: Data collection involved a lot of trekking

3.6. Statistical Analysis

Post-processing image interpretation involved textual outputs in summary sheets and contingency tables in Microsoft Excel® (MS) spreadsheet. A generic summary spreadsheet with all the input data was compiled in MS Excel. Simple statistical analyses were also conducted using MS Excel. The annual rate of change was calculated using a Declining-balance method (or Exponential Decay/Reducing balance method) formula:

Annual Land Cover Change Rate (ALCCR) = $(S1/S2)^{1/N}-1$

Where S1 = Final % Land cover, S2 = Initial % Land cover and N= Time lag in years

R statistical software (http://CRAN.R-Project.org) was used for data analysis. For the various analyses of rates of land cover change, there was aggregation of sites that were forests, altered vegetation (i.e. mosaic, agric, tree crop or urban) as one variable between T2 and T4. This was calculated as a proportion change (i.e. [(T4-T2)/T2)] and divided by a square root time difference between T4 and T2. Herbaceous, shrub land, water, flooded vegetation were treated as separate variables.

Regression, checking of quadratics and multiple regressions were used to determine the significant ($P<0.05$) relationship between correlates. Parameter estimates for the model and their level of significance are tabulated for each correlation. A P- value of less than 0.05 implies that the parameter is significant to the model. The Multiple R Squared is the coefficient of multiple determinations, which indicates the percentage contribution of the model to understanding the relationship between the variables. Parameter estimates are quoted with Standard Error (SE). All means are presented by ±SE. For statistical methods and analysis, Zar (2010) was used for reference.

CHAPTER FOUR: RESULTS AND DISCUSSION

This chapter is organised by sub-sections, the first presents accuracy assessment analysis, the second provides a broad patterns of land cover change and each of the next four representing themes or study questions. The final sub-section presents overall discussions based on results from preceding sub-sections. For each study question, there is a stand-alone sub-section covering introduction, aims of the relevant study question, methods used and analyses done, results and discussion of the results. A general discussion of the overall results is made at the end of the chapter. It is envisioned that formatting and organizing the chapter in this manner will significantly simplify the next stage of publishing of the results in a chapter by chapter approach.

4.1. Accuracy Assessment

4.1.1. Introduction

Remotely sensed data will be useful and effective when appropriate technique of accuracy assessment is performed (Senseman *et al.,* 1995*)*. Accuracy assessment is a comparison of a map produced from remote sensed data (e.g. output data from Land Cover Change Tool) with a map from validation data and thus determines the quality of information derived from remotely sensed data (Congalton and Green, 1999). Ground truth data are crucial for classifying the data to informational classes and for assessing classification accuracy. Classification accuracy is expressed by the overall accuracy percentage computed from the sum of the error, confusion, or, miscalculation matrix resulting from the application of a classifier (Fitzgerald and Less, 1994). Basic approach to accuracy assessment is the confusion or error matrix (Strahler *et al.*, 2006).

4.1.2. Aims

The main objective of accuracy assessment is to derive a quantitative description of the accuracy of the image interpretation and analysis.

4.1.3. Methods

Reference data was collected using the same land cover classification scheme used to during the land cover classification map. A total of 457 sample points was used for the accuracy assessment and ground-truthed data was assumed to be 100% correct. The accuracy assessments were computed by importing all the 1,565 ground-truthed reference points (samples) from the 20 sites into ArcMap and overlaying the points to the 2007-2008 satellite images for each site. These images were used to because the dates are as close as possible to the time when ground truth data was acquired. Each point was buffered by a radius of 100m within which the interpretation was confined. Interpretation of dominant land cover classes on the remotely-sensed data was done independent from the ground-truth data. This was summarised in a confusion/error matrix developed capturing all the 11 land cover classes and the sites. Evaluation of the accuracy of classification was computed.

Overall accuracy indicates the probability of point in the satellite image to be correctly classified within a certain degree of confidence. The total accuracy was calculated by dividing the number of correctly classified samples with the total number of samples. However, since the total accuracy value is an average, it does not reveal if the error was evenly distributed between classes or if some classes were really bad and some good and hence the need to use the users and producer's accuracy statistics. The user accuracy indicates the probability of any given sample point in any class in the satellite image to have been mapped correctly in reference to the ground truth data. This was calculated by dividing the number of correctly mapped sample points in each class with the total number of points in that class. The Producer accuracy indicates the probability that a randomly selected point on the ground truth data being mapped correctly on the satellite image. This was calculated by the number of correctly mapped points per class with the total number of points in each class in the ground truth map. Mean accuracy is a combination of user and producer accuracy.

The Kappa Coefficient of Agreement reflects the difference between actual agreement and the agreement expected by chance, between two maps (e.g. output map of classification and ground-truthed map)". The overall Kappa computed for all classes in the map indicates whether a map is below, equal or above a random agreement. A value of -1 indicates no agreement, 0 indicates a random agreement while a value of 1 depicts perfect agreement. Therefore a map with a Kappa value of 1 indicates the map and the validation data have the same values. Whereas a kappa of -1 indicates that the map classes does not correspond with the validation data at all.

4.1.4. Results

An overall interpretation classification accuracy of about 72.6% was obtained when all land classes were considered separately (Table 9). However, this increased slightly to 77% when similar categories (e.g. dense forest and open forest; agriculture-arable and natural-agricultural mosaic) were merged (Table 10). For the 11 land cover classes, the Kappa statistic results of 0.68 means that there is a 68% better agreement than by chance alone. The results user's accuracy ranged from 57% and 100% whereas the producer's accuracy ranged from 48% and 100%. The results are presented in Table 9.

Table 9: Accuracy assessment report of Landsat images for the 2006-2008 period in 20 IBA sites in Kenya and Uganda (with classes not merged)

		Groundtruth data											
	Land cover class	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**Total (C)**
Remotely sensed data	**1**	**162**	11	0	0	0	3	0	0	0	0	0	176
	2	22	**120**	12	41	0	7	0	0	0	0	0	202
	3	0	19	**104**	11	28	0	21	0	0	0	0	183
	4	0		0	**76**	51	0	5	0	0	0	0	132
	5	0		17	0	**149**	0	26	0	15	1	0	208
	6	0		11	6	0	**79**	7	0	0	0	0	103
	7	0		12	0	82	13	**326**	0	3	2	0	438
	8	0		0	0	0	0	0	**3**	3	0	0	6
	9	0		0	0	0	0	0	0	**109**	0	0	109
	10	0		0	0	0	0	0	0	0	**7**	0	7
	11	0		0	0	0	0	0	0	0	0	**1**	1
	Total (B)	184	150	156	134	310	102	385	3	130	10	1	1,565
	Total (B*C)	32,384	30,300	28,548	17,688	64,480	10,506	168630	18	14,170	70	1	366,795
	Users accuracy	0.92	0.59	0.57	0.58	0.72	0.77	0.74	0.50	1.00	1.00	1.00	
	Producers accuracy	0.88	0.80	0.67	0.57	0.48	0.77	0.85	1.00	0.84	0.70	1.00	
	Mean accuracy	0.90	0.68	0.61	0.57	0.58	0.77	0.79	0.67	0.91	0.82	1.00	
	Total correctly mapped samples	1136											
	Kappa Coefficient	0.91	0.55	0.52	0.54	0.65	0.75	0.67	0.5	1.00	1.0	1.00	
	Overall Kappa coefficient	0.68											
	Overall Accuracy	72.6%											

Land Cover classes:

1. Dense forest
2. Open forest
3. Natural/agric mosaic
4. Shrub land
5. Grassland
6. Agric – Plantation
7. Agriculture - arable
8. Open water
9. Flooded vegetation/wetland
10. Urban
11. Bare

Table 10: Accuracy assessment report of Landsat images for the 2006-2008 period in 20 IBA sites in Kenya and Uganda (with classes merged)

		Groundtruth data									
Remote sensing data	Land cover type	1	2	3	4	5	6	7	8	9	Total (C)
	1	315	41	0	10	12	0	0	0	0	378
	2	0	76	51	0	5	0	0	0	0	132
	3	0	0	149	0	43	0	15	1	0	208
	4	0	6	0	79	18	0	0	0	0	103
	5	19	11	110	13	463	0	3	2	0	621
	6	0	0	0	0	0	3	3	0	0	6
	7	0	0	0	0	0	0	109	0	0	109
	8	0	0	0	0	0	0	0	7	0	7
	9	0	0	0	0	0	0	0	0	1	1
	Total (B)	334	134	310	102	541	3	130	10	1	1,565
	Total (B*C)	126,252	17,688	64,480	10,506	335,961	18	14,170	70	1	569,146
	Users accuracy	0.83	0.58	0.72	0.77	0.75	0.50	1.00	1.00	1	
	Producers accuracy	0.94	0.57	0.48	0.77	0.86	1.00	0.84	0.70	1	
	Mean accuracy	0.88	0.57	0.58	0.77	0.80	0.67	0.91	0.82	1	
	Total correctly mapped samples	1202									
	Kappa Coefficient	0.002	0.004	0.002	0.007	0.001	0.111	0.007	0.099	1	
	Overall Kappa coefficient	0.70									
	Overall Accuracy	76.8%									

Land Cover classes (when merged):

1. Forest
2. Shrub land
3. Grassland
4. Agric – Plantation
5. Agriculture - arable
6. Open water
7. Flooded vegetation/wetland
8. Urban
9. Bare

4.1.5. Discussion

The results indicate that the total accuracy was 72.6% whereas the user's and producer's accuracy ranged from 57% and 100% and 48% and 100% respectively. The overall Kappa computed for all classes in the map indicates whether the interpreted remote sensed data is below, equal or above a random agreement. Since a value of -1 indicates no agreement (i.e. interpreted data does not correspond with validation data at all), 0 indicates a random agreement and 1 depicts a perfect agreement, all values are closer to 1 and therefore an indicating a perfect agreement for flooded vegetation, urban and bare ground land cover classes and a near perfect agreement for dense forest.

The overall accuracy of 72.6% is below the 85% accuracy target widely accepted by the remote sensing community as a benchmark (Foody, 2002; Rese *et al.*, 2002; Fuller *et al.*, 2003). When similar land cover categories were merged, the overall interpretation accuracy increased. It therefore means that further merging of classes would have attained an overall accuracy closer to 85%. However, the viability of the 85% accuracy target has been questioned. Wulder *et al.*, (2006) feels that the usefulness of the 85% benchmark as a standard is unclear confirming similar doubts by Laba *et al.*, (2002) based on the argument that user's and producer's accuracies are stabilizing in the 50-70% range and that map accuracies will approach 80% only through the use of high spectral, spatial and temporal resolution images. In fact Czaplewski and Patterson (2003), for land cover product having more than 10 classes, accuracies for each stratum are required to exceed 70% in which case the overall accuracy of this study exceeds the 70% proposed accuracy target.

Nevertheless, even though this accuracy may be acceptable considering the argument to cap the target at 70%. However, if the 85% target is still valid, this low value of overall accuracy could be attributed to reference data, which was collected by two different groups of field assistants in

Kenya and Uganda and hence a possibility of slight variations in their assessment of ground-truthed data.

4.2. Land cover change on IBAs in East Africa between 1986-2008

4.2.1. Introduction

Land cover change is the largest threat to globally threatened birds (BirdLife International 2004). It is linked to the intersection of natural and human influences in environmental change (Ghosh *et al.*, 1996) and these changes are driven by heterogeneous dynamics in land use (Turner *et al.*, 1995). Land cover is the observed physical and biological cover of the earth's land (e.g. vegetation or lack of, man-made features) whereas land use describe human uses of the land for a certain purpose, or immediate actions modifying or converting natural environment (land cover) or wilderness into built environment such as fields, pastures, and settlements (UNEP, 2007). The human dimension of global environmental change is concerned with the human causes of change and the consequences of such changes (Jäger, 2001). Based on scenarios of future land use and climate, land use will result in a bigger share of biodiversity loss by 2100 than climate change (Sala *et al.*,2000) with a prediction by Chapin *et al.*,(2000) that for at least the next 100 years land cover change is likely to be the most significant variable impacting on biodiversity. In fact these changes are unprecedented, faster in the last 50 years than at any time in human history and projected to continue (Millennium Ecosystem Assessment, 2005).

Land cover and land use change is manifested through conversion of natural habitats into agriculture, pasture, urban areas and other infrastructural developments (e.g. roads). Land cover change analysis is important in identifying proportions of land cover categories that have changed over a particular period. Such analysis and mapping of change have therefore helped in understanding the dynamics and impacts of human activities in time and space. Based on this understanding appropriate interventions that target involvement of various actors in addressing threats, setting priorities, designing interventions approaches, and conservation investments are bolstered, setting in motion monitoring, evaluating and adaptive management strategies and processes aimed at ameliorating the effects of deleterious land cover change.

4.2.2. Aims

This section presents the broad patterns of land change dynamics at the 72 study areas in East Africa during a 22-year study period (1986-2008). It also shows through comparisons how the various land cover categories have changed during the two decade period.

4.2.3. Methods and Analysis

Based on the image interpretation (See Chapter 3) summaries, the proportion of each of the 11 land cover types for each site in the period around 1990, 2000 and 2006 was determined. The years around 1990 were used as baseline and changes in land cover (both positive and negative) tabulated in contingency tables and percentage land cover change determined. This analysis was summarised at a broad regional scale level, national level and site level. However, only regional level and national level changes for respective land cover categories were presented. A paired t-test was used to determine if there were significant differences between observed land cover changes inside and outside IBA sites.

4.2.4. Results

Images covered the time between 1986 and 2008. Land cover change varied ranging from positive to negative within IBAs across IBAs and outside the IBAs. On average the extent of closed forest decreased on IBAs between the time periods considered both within and outside of the IBAs (see Figure 8; T test results here for comparison between before and after). There was considerable scatter around this relationship, as shown by the wide error bars, with cover on some IBAs increasing by up to 36.8 %, while it decreased by 68.2 % on others. A similar pattern was observed for closed forest outside the IBAs with a site increasing by 33.3% while it decreased by 91.7% on another site thus providing enough evidence of significant loss of closed forest patches immediately around IBAs.

There was no significant difference in the percentage of change within IBAs in comparison to the rates of change in the surrounding 20 km buffer for all land cover classes (Table 11) apart from closed forest ($t = 1.1815$, $p=0.013$) and natural-agricultural mosaic ($t = -2.2216$, $p=0.030$). By contrast, the extent of open forest did not change significantly within IBAs between 1986-2008 (Figure 9), with some of the IBAs' open forest cover increasing by 2.7% in Ethiopia, 6.8% in Kenya and 31.25% in Burundi (Tables 11, 12, 13). In contrast, there was considerable loss around the IBAs with open forest cover increasing by up to 66.7%, while it decreased by up to 42.9% on others. Changes in each of the land cover types are summarised in Figures 9-19 and Table 11. General trends show that during the period natural land cover decreased. In order of magnitude, 19.4% of closed forest, (Figure 8) 17.84% of bare ground (Figure 18), 6.23% of flooded vegetation (Figure 16), 3.37% of shrub (Figure 11) and 3.13% of area under natural-agricultural mosaic (Figure 10), were lost and converted into other land cover types such as open forest, herbaceous, arable, and urban. This is demonstrated by a proportionate increase or gain in area under urban (50.95%), arable (32.57%), herbaceous (2.26%) and open forest (1.63%) within the IBAs respectively (Table 11). The pattern suggests general conversion and alteration of natural vegetation to artificial, modified land cover types.

During the same period, land cover changes also took place within the buffer or the area surrounding the IBAs. The general trend show a loss in closed forest (-33%; Figure 8), open forest (-38.6%; Figure 9), natural-agricultural mosaic (-23.3%, Figure 10), flooded vegetation (-20.5%, Figure 16), shrub land (-11.4%, Figure 11), bare ground (-8.98%, Figure 18) and herbaceous (-3.9%, Figure 12) respectively. These appear to have been replaced by altered landscapes which increased in proportion, with urban up 72.8% (Figure 17), arable up 24.1% (Figure 14) and, Tree/shrub crops up 9.7% (Figure 13). All these changes and rates of change during the 22 year period are presented in Table 11.

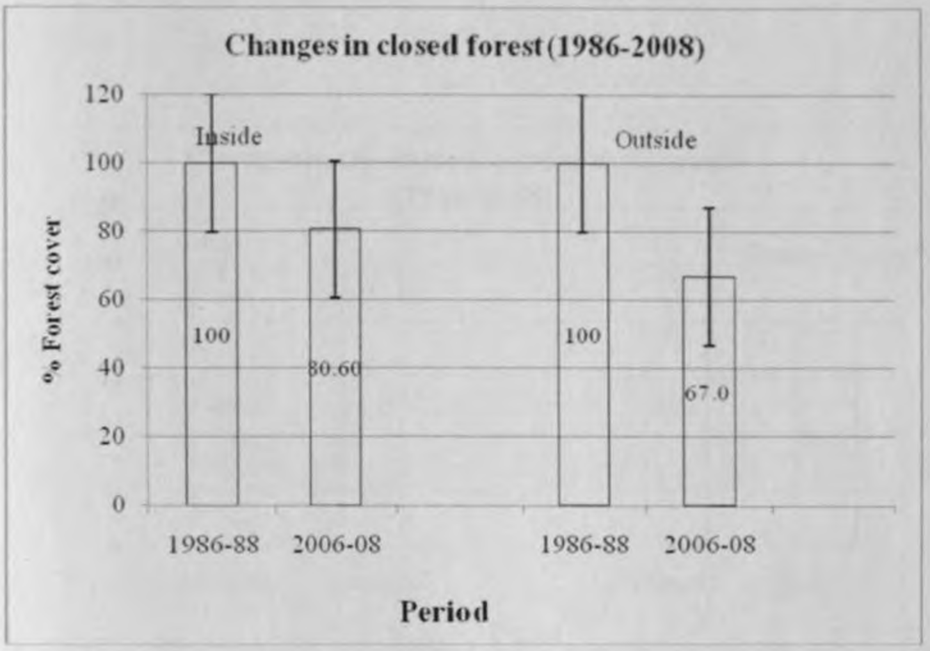

Figure 8: Closed forest land cover change during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda n=28)

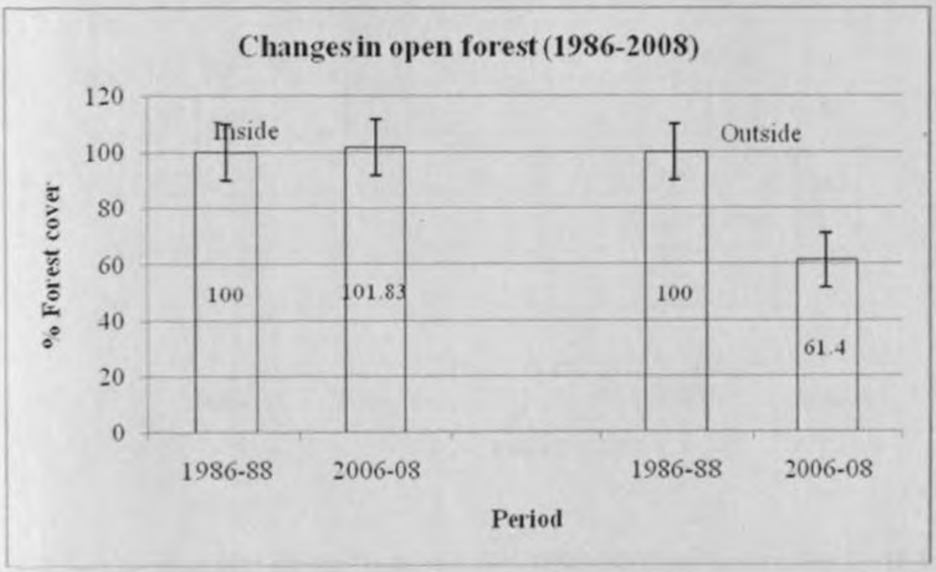

Figure 9: Open forest land cover change during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda, n=44)

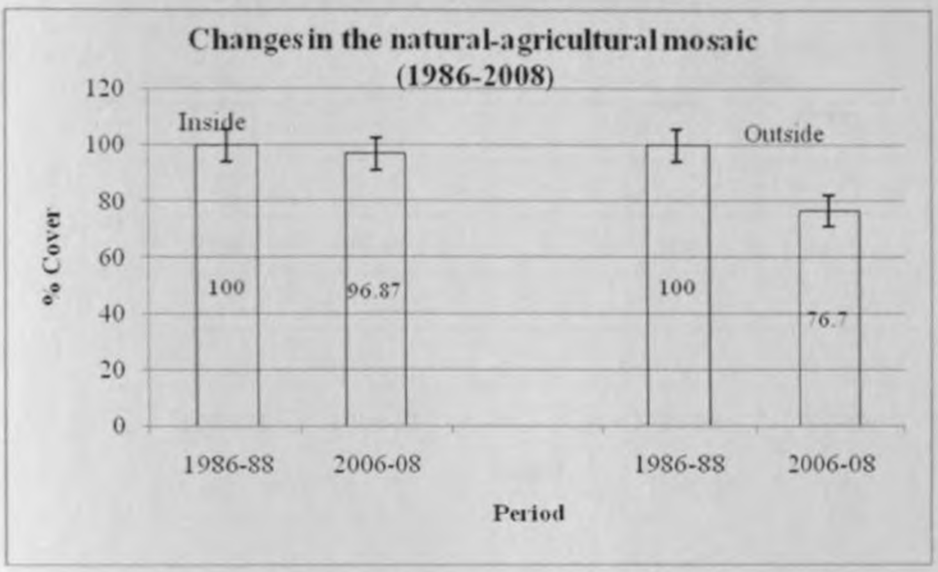

Figure 10: Land cover change in the natural-agricultural mosaic cover during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda n=48)

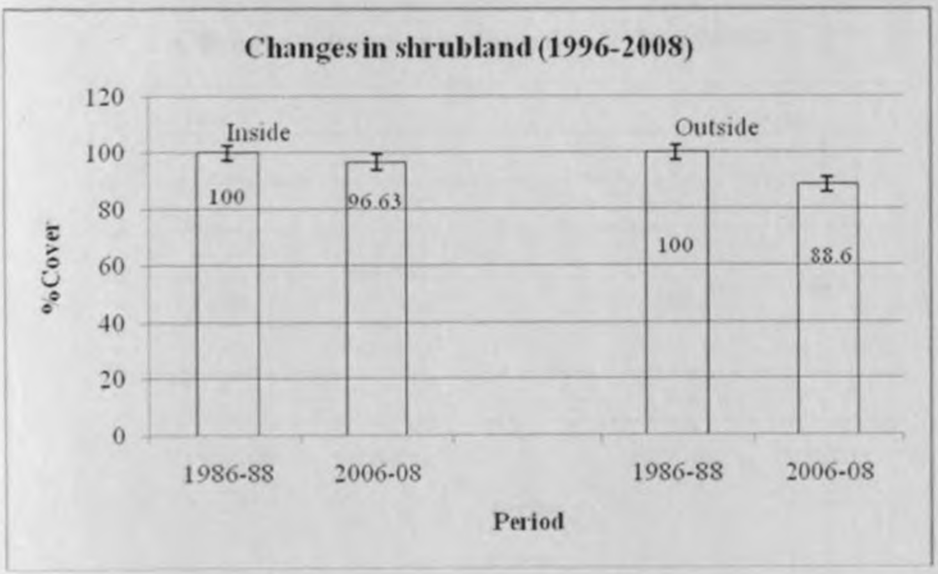

Figure 11: Shrub land cover change during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda n=56)

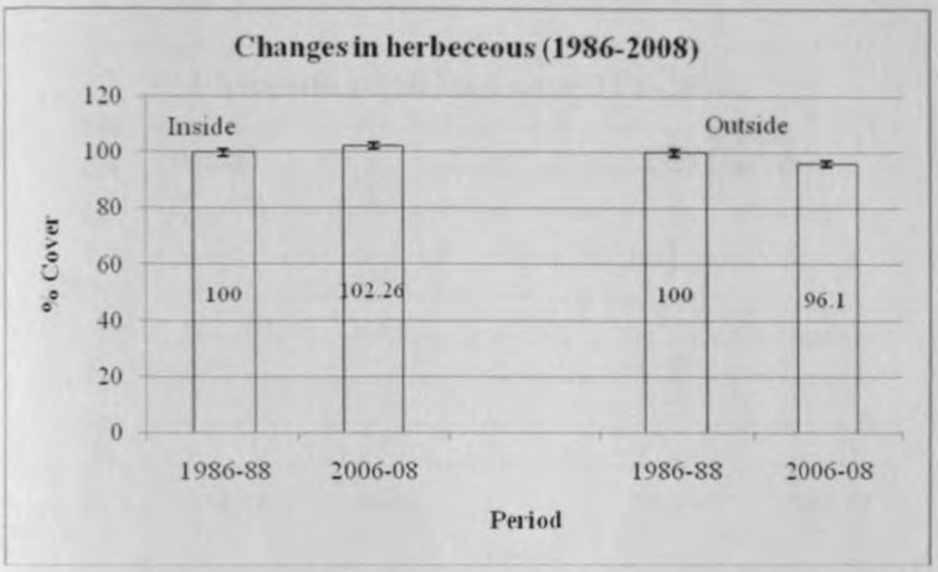

Figure 12: Herbaceous land cover change during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda n=53)

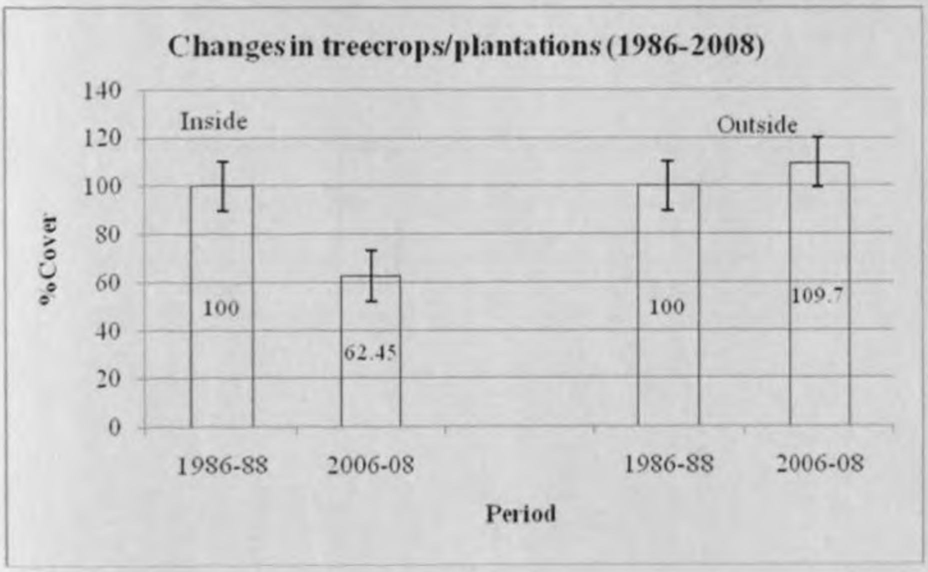

Figure 13: Tree/shrub crop land cover change during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda n=22)

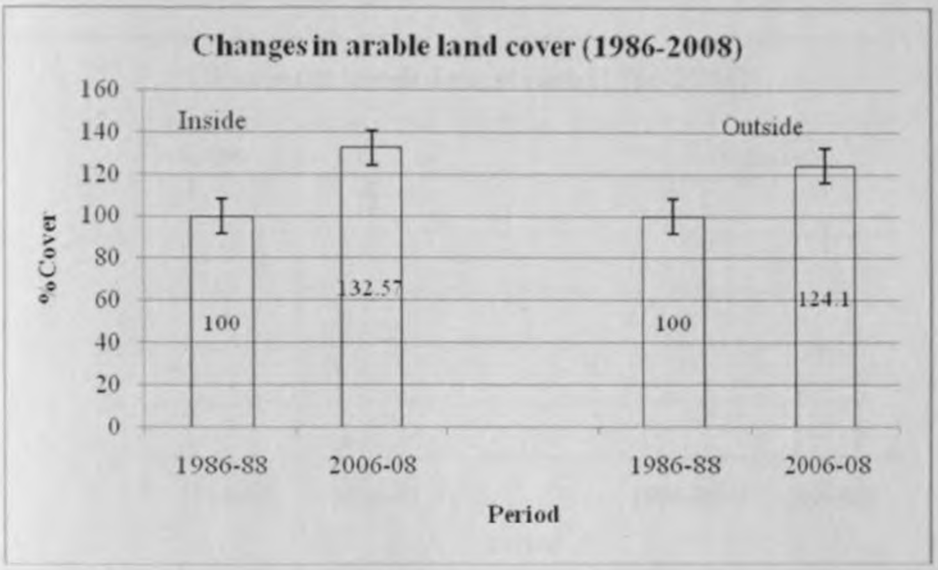

Figure 14: Arable land cover change during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda n=51)

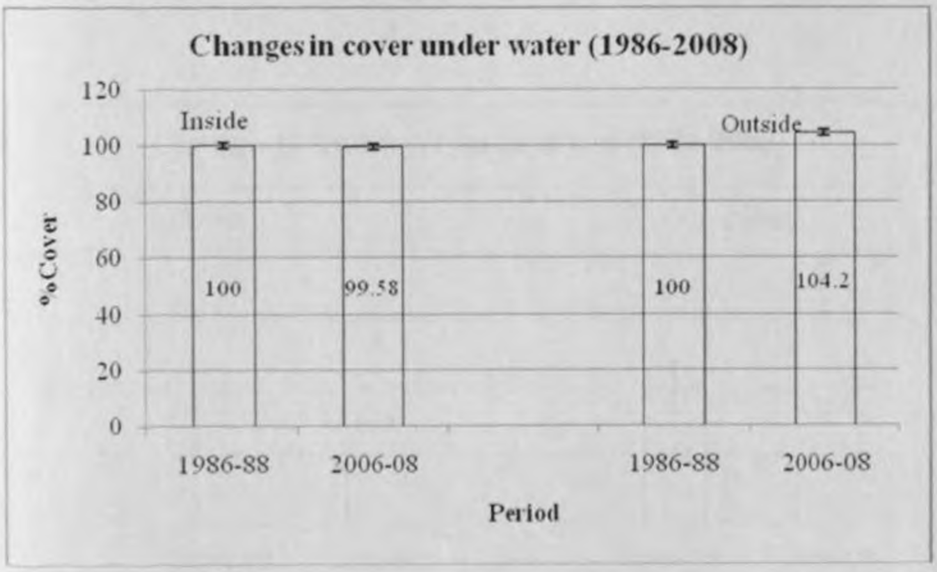

Figure 15: Land cover change for water during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda, n=27)

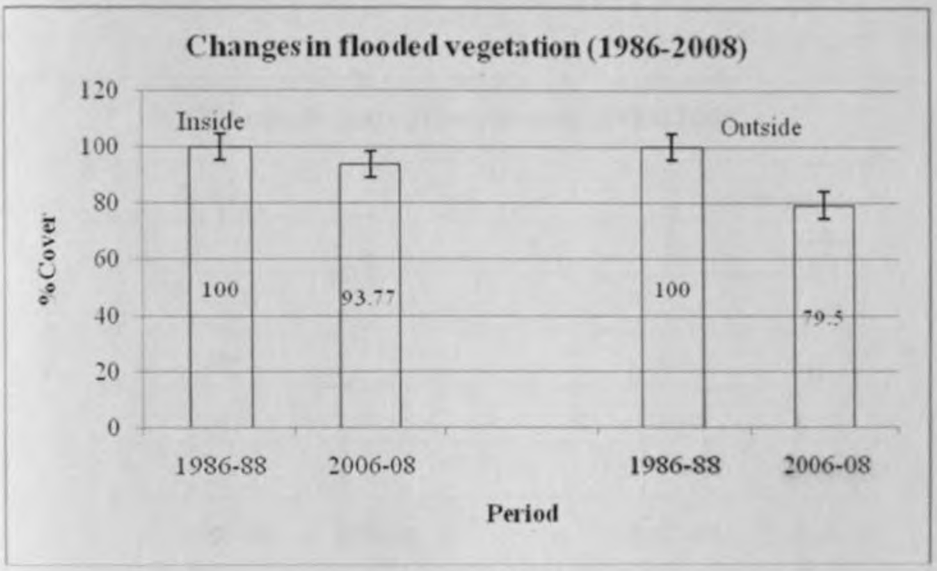

Figure 16: Flooded vegetation: land cover change during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda, n=25)

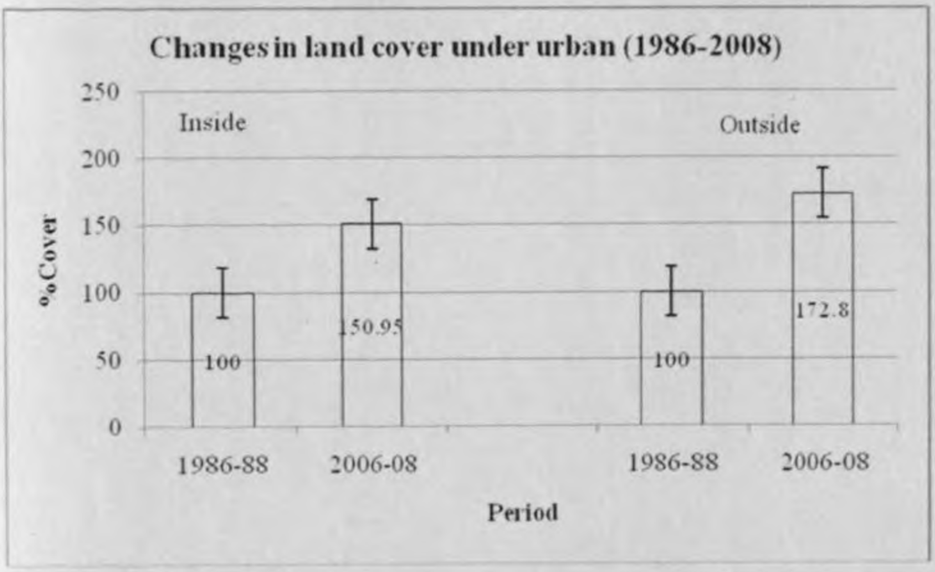

Figure 17: Urban: land cover change during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda, n=5)

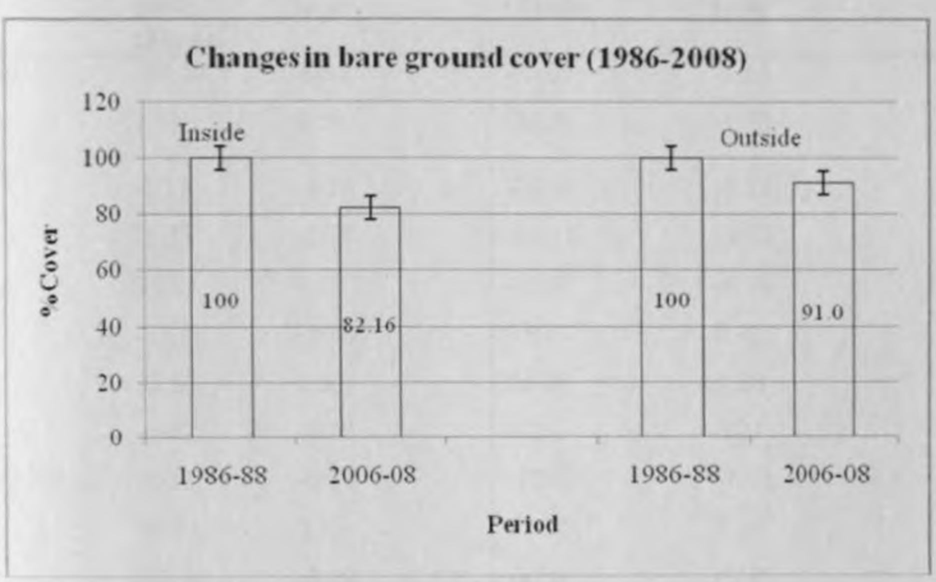

Figure 18: Bare ground land cover change during the period 1986-2008 both inside and outside the IBA for IBAs within the Eastern Africa region (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda, n=18)

Table 11. Summary table showing the percentage land cover change at regional level per land cover type and the estimated rates of change over a 20 year period.

	Inside		**Outside**		**T-Test**
Land Cover Type	**% Land cover change**	**Rate of change (p.a)**	**% Land cover change**	**Rate of change (p.a)**	**P-Value**
Closed forest	-19.40	-0.97	-33.00	-1.65	**0.013**
Open forest	1.83	0.09	-38.58	-1.93	0.319
Natural-agric mosaic	-3.13	-0.16	-23.34	-1.17	**0.030**
Shrub	-3.37	-0.17	-11.37	-0.57	0.112
Herbaceous	2.26	0.11	-3.88	-0.19	0.828
Tree crop	-37.55	-1.88	9.67	0.48	0.676
Arable	32.57	1.63	24.06	1.20	0.746
Water	-0.42	-0.02	4.25	0.21	0.194
Flooded vegetation	-6.23	-0.31	-20.52	-1.03	0.986
Urban	50.95	2.55	72.77	3.64	0.416
Bare	-17.84	-0.89	-8.96	-0.45	0.712
Mean	**-0.030**	**-0.002**	**-2.627**	**-0.133**	
STD Error	**7.28**	**0.36**	**9.38**	**0.47**	

At national level (Burundi, Ethiopia, Kenya, Rwanda, Tanzania and Uganda), trends show a general decrease in closed forest land cover in all the countries ranging from 11-60% (-11.9% in Kenya, -15.4% in Uganda, -25.2% in Burundi, -39.3% in Burundi and -60% in Ethiopia, Tables 12-16). Natural-agricultural mosaic land cover increased in size but decreased in other countries. For example, it increased by 18.7% in Tanzania and 55.7% in Burundi but at the same time decreased by 9.7% in Kenya, 10.8% in Uganda and 32.3% in Ethiopia (Tables 14, 16 and 13). The area under shrub land increased by 1.4% in Kenya, 2.8% in Ethiopia and 42.8% in Uganda but decreased by 13.9% in Burundi and 18.8% in Tanzania) and flooded vegetation (Tables 12-16). Again when considering results from land cover change analysis outside the forest, similar trends are observed (Tables 12-16) in terms of loss of natural vegetation (closed forest, open forest, shrub land, herbaceous and flooded vegetation) and increase in the area under modified/altered land cover types (arable, plantation crops, urban) as presented in Tables 12-16.

At the country level, when comparing whether there was a significant difference between land cover change inside and outside the IBAs, there is no significant difference for all land cover types in Burundi apart from natural-agricultural mosaic (Table 12). In Ethiopia, the trend is almost the same such that there is no significant difference for all land cover types considered in this analysis (Table 13). In contrast, even though not significant for the majority of land cover types, there is a significant difference in land cover change inside and outside IBAs for closed forest in Kenya (Table 14) and Tanzania (Table 15) and for open forest and water in Uganda (Table 16).

Table 12: Summary table showing the percentage land cover change in Burundi IBAs per land cover class and the estimated rates of change over a 20 year period.

	Inside the IBAs		**Outside the IBAs**		**T-Test**
Land Cover Type	**% Land cover change**	**Rate of change (p.a)**	**% Land cover change**	**Rate of change (p.a)**	**P-Value**
Closed forest	-39.26	-2.0	-9.6	-0.5	-
Open forest	31.25	1.6	-9.3	-0.5	0.759
Natural-agric mosaic	55.69	2.8	-25.0	-1.2	**0.035**
Shrub	-13.91	-0.7	-0.9	0.0	0.711
Herbaceous	6.09	0.3	-9.1	-0.5	0.728
Tree crop	-	-	209.1	10.5	-
Arable	47.14	2.4	40.0	2.0	0.312
Water	-	-	-36.8	-1.8	-
Flooded vegetation	-2.90	-0.1	14.1	0.7	0.487
Urban	-	-	-28.5	-1.4	-
Bare	0.00	0.0	100.0	5.0	-
Mean	**10.513**	**0.538**	**22.182**	**1.118**	
STD Error	**11.35**	**0.57**	**21.98**	**1.10**	

Table 13: Summary table showing the percentage land cover change in Ethiopia per land cover type and the estimated rates of change over a 20 year period.

	Inside		Outside		T-Test
Land Cover Type	**% Land cover change**	**Rate of change (p.a)**	**% Land cover change**	**Rate of change (p.a)**	**P-value**
Closed forest	-60	-3	-19.0	-0.9	-
Open forest	2.7	0.1	4.6	0.2	0.8868
Natural-agric mosaic	-32.3	-1.6	-62.9	-3.1	0.1933
Shrub	2.8	0.1	-14.0	-0.7	0.6377
Herbaceous	21.5	1.1	5.3	0.3	0.5213
Tree crop	-42.9	-2.1	-18.5	-0.9	-
Arable	18.5	0.9	16.3	0.8	0.5412
Water	-20.9	-1.0	14.2	0.7	0.7397
Flooded vegetation	-88.5	-4.4	-87.6	-4.4	0.7688
Urban	-0.4	0.0	21.4	1.1	0.6879
Bare	0.2	0.0	-89.3	-4.5	0.1853
Mean	**-18.118**	**-0.9**	**-20.864**	**-1.036**	
STD Error	**10.42**	**0.52**	**15.32**	**0.62**	

Table 14: Summary table showing the percentage land cover change in Kenya per land cover type and the estimated rates of change over a 20 year period.

	Inside		Outside		T-Test
Land Cover Type	**% LCC**	**Rate LCC (p.a)**	**% LCC**	**Rate of LCC (p.a)**	**P value**
Closed forest	-11.9	-0.6	-62.6	-3.1	**0.026**
Open forest	6.8	0.3	-42.3	-2.1	0.977
Natural-agric mosaic	-9.7	-0.5	-29.0	-1.5	0.244
Shrub	1.4	0.1	-18.9	-0.9	0.267
Herbaceous	-9.4	-0.5	-10.8	-0.5	0.982
Tree crop	9.5	0.5	9.0	0.4	0.273
Arable	21.1	1.1	25.8	1.3	0.744
Water	0.9	0.0	8.3	0.4	0.374
Flooded vegetation	5.7	0.3	-1.4	-0.1	0.311
Urban	216.0	10.8	224.6	11.2	0.595
Bare	-5.9	-0.3	13.3	0.7	0.59
Mean	**20.409**	**1.018**	**10.545**	**0.527**	
STD Error	**19.78**	**1.00**	**22.81**	**1.14**	

Table 15: Summary table showing the percentage land cover change in Tanzania per land cover type and the estimated rates of change over a 20 year period.

	Inside		Outside		T-Test
Land Cover Type	**% Land cover change**	**Rate of change (p.a)**	**% Land cover change**	**Rate of change (p.a)**	**P value**
Closed forest	-25.2	-1.3	-37.1	-1.86	**0.036**
Open forest	-8.2	-0.4	-47.6	-2.38	0.879
Natural-agric mosaic	18.7	0.9	4.1	0.21	0.980
Shrub	-18.8	-0.9	-13.5	-0.68	0.272
Herbaceous	-6.7	-0.3	-3.0	-0.15	0.383
Tree crop	-66.9	-3.3	-26.9	-1.34	0.200
Arable	64.7	3.2	63.3	3.16	0.376
Water	20.8	1.0	6.8	0.34	0.304
Flooded vegetation	-22.3	-1.1	6.4	0.32	0.925
Urban	90.1	4.5	61.4	3.07	0.805
Bare	-43.6	-2.2	-50.1	-2.51	0.262
Mean	**0.236**	**0.009**	**-3.291**	**-0.165**	
STD Error	**13.84**	**0.69**	**11.61**	**0.58**	

Table 16: Summary table showing the percentage land cover change in Uganda per land cover type and the estimated rates of change over a 20 year period.

	Inside		Outside		T-Test
Land Cover Type	**% Land cover change**	**Rate of change (p.a)**	**% Land cover change**	**Rate of change (p.a)**	**P-value**
Closed forest	-15.4	-0.8	-23.0	-1.1	0.579
Open forest	-4.6	-0.2	-35.3	-1.8	**0.093**
Natural-agric					0.316
mosaic	-10.8	-0.5	-32.4	-1.6	
Shrub	42.8	2.1	4.1	0.2	0.526
Herbaceous	5.6	0.3	-0.5	0.0	0.690
Tree crop	18.0	0.9	30.3	1.5	0.316
Arable	62.4	3.1	7.8	0.4	0.367
Water	5.0	0.3	6.3	0.3	**0.088**
Flooded vegetation	-18.2	-0.9	-16.6	-0.8	0.275
Urban	-	-	69.6	3.5	-
Bare	-43.1	-2.2	-12.4	-0.6	0.546
Mean	**4.170**	**0.210**	**-0.191**	**0.000**	
STD Error	**9.26**	**0.49**	**9.11**	**0.46**	

However, there were no significant differences between countries in change in dense forest land cover, open forest, mosaics, shrub, herbaceous water urban or bare (Table 17). Differences were detected in rate of change for tree/shrub crops (plantations) land cover type, with greater loss in Tanzania than in Ethiopia. There was a trend for change in tree crops, arable and flooded vegetation to differ across the countries studied (Table 17). For example, there was a greater gain in arable land in Tanzania (64.7%), Uganda (62.4%), Burundi (47.1%), Kenya (21.1%) and Ethiopia (18.5%).

At a regional scale, there was a significant difference between changes in open forest (including woodlands) in and outside IBAs across the countries.

Table 17: Comparisons of land cover change (1986-2008) between the various countries in the Eastern Africa region (within the IBAs)

	Std. Error	R^2	F-Statistic	Pr (>|t|)
Closed forest	0.1899	-0.02289	$F_{5, 28}=0.8523$	0.5248
Open forest	0.2271	-0.06425	$F_{5, 44}=0.4083$	0.8404
Nat-agric mosaic	0.3182	0.05194	$F_{5, 48}=1.581$	0.1835
Shrub land	0.2632	-0.03239	$F_{5, 56}=0.6172$	0.6871
Herbaceous	0.136	-0.05729	$F_{5, 53}=0.3715$	0.866
Tree/shrub crop	0.2368	0.3899	$F_{5, 22}=6.326$	**0.0029**
Arable	0.2291	0.1017	$F_{5, 51}=2.268$	**0.0615**
Water	0.0875	0.0697	$F_{5, 27}=1.58$	0.2081
Flooded veg.	0.2808	0.1613	$F_{5, 25}=2.154$	**0.0918**
Urban	0.5889	-0.181	$F_{2, 5}=0.4634$	0.6537
Bare ground	0.4115	0.0481	$F_{4, 18}= 1.278$	0.3153

Table 18: T-Test results showing the level of significance of land cover changes (1986-2008) inside and outside IBAs at a regional level

Inside IBAs				Outside IBAs		
Inside IBAs	**T value**	**DF**	**P Value**	**T value**	**DF**	**P Value**
Closed forest	0.812	136.790	0.418	1.231	128.849	0.220
Open forest	-0.086	137.408	0.932	2.091	116.868	**0.039**
Nat-agric mosaic	0.125	135.805	0.901	1.516	132.019	0.132
Shrub land	0.165	135.965	0.869	0.662	135.884	0.509
Herbaceous	-0.132	135.718	0.896	0.266	135.686	0.791
Tree/shrub crop	0.771	101.411	0.443	-0.220	135.686	0.826
Arable	1.041	128.928	0.300	-1.488	133.129	0.139
Water	0.014	135.857	0.989	-0.116	135.228	0.908
Flooded veg.	0.173	135.576	0.863	1.094	122.411	0.276
Urban	-0.584	120.253	0.560	-0.584	120.253	0.560
Bare ground	0.280	135.907	0.780	0.219	132.608	0.827

4.2.5. Discussion

At the regional level, analysis of the 72 IBAs provides an overview of the basic patterns of change, average loss of land cover and the mean rates of change both within and around IBAs during a time covering between around 1986 and 2006. The general trend shows a decline in closed forest, natural agricultural mosaic, shrub land, tree/shrub crop, flooded vegetation and bare ground cover within IBAs. In general, there appears to be a general decline in the area under natural vegetation within the IBAs (e.g. closed forest, natural-agricultural mosaic, shrub land, water, flooded vegetation) and increase in the area under herbaceous, arable, urban (i.e. human encroachment and settlement) as presented in Figures 8-18.

As an indicator of the reducing land cover e.g. closed forest, there was an increase in open forest probably as a result of opening up of the closed forest through deforestation, selective logging, encroachment (i.e. increase in area covered by urban) and arable within the IBAs. For example, at a regional level, open forest increased by 1.83% while arable land increased by 32.6% and urban by 50.95% even though this varied across the countries covered in this study.

Generally, these findings are consistent with FAO (2009), which points to a reducing forest cover globally even though they may not actually coincide. The area under herbaceous cover, arable and urban (manifested through human settlement) increased during the same period. This could be evidence that natural habitats were transformed into altered habitats characterised by agricultural and human settlement landscapes. The general trend shows a decline in natural land cover types and though not of the same magnitude, the downward trends are consistent with a study by Olson and Maitima (2006), which shows an expansion of agriculture at the expense of grazing land in East Africa over the past 50years.

Flooded vegetation which is mainly composed of hydrophytic vegetation is the most visible and easily recognisable diagnostic feature of wetlands and wetlands change (Munyati, 2000) and therefore a decline in its coverage translates into a decline in wetlands cover with implications on the survival across the range of species restricted to this habitat type. This study presents a scenario of wetland ecosystems increasingly under threat from the direct effects of conversions and modifications by human or a run of dry years. In a study by Owino and Ryan (2007), papyrus (hydrophytic/flooded vegetation) cover was lost by 50% (Dunga), 47% (Koguta) and 34% (Kusa) respectively between 1969 and 2000. These human activities such as urbanization, farming, horticulture (e.g. green houses) was very evident during the image interpretation analysis as well as ground-truthing field work at some of the sites visited (e.g. land transformations and conversions at Yala Swamp and Lutembe Bay IBAs in Kenya and Uganda respectively). However, on the contrary, the open forest land cover reduced outside the IBAs during the same period. This could be as a result of the modification of existing natural habitats outside IBAs and conversion to other forms of land use plus any small forest patches outside IBAs.

Similar trends are observed outside the IBAs where the area under natural vegetation decreased in order of increasing magnitude for open forest (-38.6%), closed forest (-33%), natural-agricultural mosaic (-23.3%), flooded vegetation (-20.52), shrub land (-11.4%), bare (-8.96%) and herbaceous (-3.9%). However, during the same period, agricultural intensity increased by 24.06% outside IBAs just tree/shrub crop (9.7%), arable, water (4.3%) and urban (including human settlement increased by 73.8% outside the IBAs.

A decline in each of the land cover types within IBAs could have an implication on the conservation status of the various species dependent on these sites. For example forest, shrub land (e.g. Hinde's babbler) and wetland dependent species could have been affected most because of reduced cover in their preferred habitats. Agricultural expansion and deforestation are the most prevalent threats to IBAs in Africa (Buchanan *et al.,* 2009a), and the results presented here suggest that they are the two most important land cover related changes that are affecting IBAs. A study by DeFries *et al.,* (2010) revealed the growing influence on urbanisation on protected areas even though in that study, urbanisation was only associated with the emergence of towns and not just human settlement around key biodiversity sites and could not document whether urbanisation has a positive or negative influence on conservation.

Many factors could be driving these changes. For example, based on Slingenberg *et al.*, (2009), direct causes of forest ecosystem changes in Tanzania included large scale deforestation, agricultural expansion, wood extraction, illegal logging, and mining, while underlying causes of ecosystem change include population growth, inadequate quality control mechanisms, and lack of umbrella environmental legislation among other factors. Understanding the relative influence of each of these direct and underlying factors across the sites in the region is therefore critical in formulating strategies to ensure effective interventions. It can also be argued that the differences in the extent of adverse land cover change in the respective countries is as a result of the different management regimes and practices at national level. The cumulative impact of this is that these differences in approaches (management styles, systems, and effectiveness, farming systems) either accelerate or reduce deleterious land cover change.

4.3. Pressures on IBAs (i): Is the loss of natural land cover related to the human population density around IBAs

4.3.1. Introduction

The state of human populations and the environment are inextricably linked (WWF Conservation Strategies Unit, 2002). Human population growth and the increased demand and ludicrous per capita consumption of natural resources are factors driving global environmental change. For example, the underlying conditions determining the type and intensity of human activities (overconsumption, overexploitation, demand for food and shelter) lead to biodiversity loss (Kerr and Currie, 1995; Vitousek *et al.*, 1997; Sala *et al.*, 2000; Engelman *et al.*, 2000). The combination of increasing human numbers and consumption, together with human land use patterns, is changing the planet on a scale and with a speed that is unprecedented (WWF Conservation Strategies Unit, 2002). Human population density is one of the measures of the distribution of threat besides habitat loss (Kerr and Currie, 1995; Thompson and Jones, 1999; McKinney, 2001; Brooks *et al.*, 2002; Scharlemann *et al.*, 2005).

The fact that various studies have shown that human population density is positively correlated with terrestrial vertebrate and plant species richness in Africa (Balmford *et al.*,2001a, b) and partial correlation in Europe (Araújo, 2003) is a cause for concern. For example, more than 1.1 billion people live in areas that conservationists consider the richest in non-human species and the most threatened by human activities (Engelman *et al.*, 2000). Threats from human activities are largely responsible for the current rate of species loss, and pressure on habitats and species will continue to proportionately increase given predicted growth rates of human populations (UNPD, 2003).This therefore means that if population density as a factor could be driving land cover change at these sites of species richness, this could have very serious ramifications on the sites and species.

Yet population is still growing rapidly. At the global level, human population is predicted to rise to 9 billion by 2050 (Lutz *et al.*, 2001; UNPD, 2003). In Africa alone, the human population increased from 364,132,000 in 1980 to 820,959,000 in 2000 (UNEP, 2008) and is projected to double in size between 2008 and 2050 (Population Reference Bureau, 2008). Two-thirds of the population of sub-Saharan Africa lives in rural areas, and therefore depends on the natural resource base more than people in any other part of the world (Kagwanja and Dione, 2005). Human population density and growth is known to drive urbanization (WWF Conservation Strategies Unit, 2002), overconsumption and agricultural expansion (Brink and Eva, 2009).

This growth and increase in density will concomitantly increase per capita consumption (Myers and Kent, 2003) as the overall food demand is expected to increase two- to threefold by 2050 (Green *et al.*, 2005) supporting the opinion that human population growth is one of the major threats to biodiversity (Gehrt, 1996). Population density will put even more pressure on natural ecosystems (Pain *et al.*, 2005) as a result of household dynamics on resource consumption on biodiversity (Liu *et al.*, 2001). The most important agent of change in the spatial patterns of much of biodiversity at present is ultimately the size, growth and resource demands of the human population (Vitousek *et al.*, 1997; Sala *et al.*, 2000). Besides changes in spatial patterns, historical extinctions have been attributed to human activities (Soulè, 1983) while human population density has contributed to deforestation in tropical countries (Meyer and Turner, 1992; Cohen, 1997), decline in birds and mammal species (Mckinney, 2001) and predict the extinction of plants (Thompson and Jones, 1999).

Many studies have explored the relationships between human population density or growth and biodiversity change. Whereas some studies suggest that there is increased population growth near areas of high biodiversity values (e.g. Balmford *et al.*, 2001a, Scharlemann *et al.*,2005, Luck, 2007), other studies point out that human population growth accelerates near and far away from protected areas (Wittemyer *et al.*, 2008; Joppa *et al.*, 2009a). Even though there could be different methodological issues in between Wittemyer *et al.*, (2008) and Joppa *et al.*, (2009a), their point of convergence is that disproportionate increases in population growth near protected area boundaries may threaten their ability to conserve biodiversity. According to Harcourt and Sparks (2003), human population density in the geographic range of threatened primate species is significantly higher than in the range of Lower Risk species. Other studies however associate threats to biodiversity with high population density by exploring the role of population factors (e.g. migration pressure) in land use change (Angelsen and Kaimowitz, 1999; Engelman *et al.*, 2000; FAO, 2005 FAO 2009).

In addition to the impact of human population on the defined sites of conservation importance, the fragmentation of natural habitat in the landscape may also reduce populations of species and species richness (Lovejoy *et al.*, 1986; Bierregaard *et al.*, 1992; Fahrig, 1997; Hargis *et al.*, 1999; Chittibabu and Parthasarathy, 2000; Kurki *et al.*, 2000; Fahrig, 2002; Jha *et al.*, 2005). Sites are often increasingly isolated (Skole and Tucker, 1993; Petit and Lambin, 2001; Achard *et al.*, 2002) as the extent of habitat linking and surrounding the sites decreases (Bennet, 2003).

Considering the current and projected population growth, particularly in the developing world where most of the biodiversity is found, controlling population density will have a great impact on the effectiveness of reversing trends in global biodiversity loss (Slingenberg *et al.*, 2009). Pain *et al.*, (2005) recognises the need for prioritising sites for action due to current rates of population and habitat loss and increasing human numbers and demands on natural resources. Current investments by conservationists in the global biodiversity hotspots and key biodiversity sites (e.g. IBAs) are one of the interventions. Besides conservation planning and investments as an intervention, as population density increases, there is need for a continuous assessment of its impact on land cover change and biodiversity loss. Considering that human population density is positively correlated with terrestrial vertebrate and plant species richness in Africa (Balmford *et al.*, 2001a, b) and the tendency for human settlements to be located near centres of bird endemism in other parts of the world (e.g. the tropical Andes, Fjeldså and Rahbek, 1998), many key biodiversity sites (e.g. IBAs) are also surrounded by high densities of people and hence there is need to monitor the impact of population density around these sites with current rates of land cover change.

4.3.2. Aims

This section aims to test the hypothesis that land cover change is higher at IBAs surrounded by high human population densities than on sites with lower population densities. To achieve this, changes within IBAs and outside (matrix) were analysed and independently regressed with surrounding human population density. This therefore helped to assess the impact of the surrounding population on land cover change within and outside IBAs.

4.3.3. Methods and Analysis

Rates of land cover change for the 71 IBAs were determined through interpreting satellite data using the Land Cover Change Graphical User Interface (section 4.1). Human population densities (persons per square km) data at a 1 km resolution for 2005 was extrapolated from the Gridded Population of the World (GPW) database (CIESIN and CIAT, 2005).

Based on the dominant land cover type at each IBA, the sites were categorised into six broad classes namely; artificial landscape, forest, grassland, natural-agricultural mosaic, shrub land and wetlands. The potential correlations between human population density around the sites and broad land cover were investigated to determine a) if loss of natural vegetation was higher in areas of greater human population density, and b) if this relationship differed between different dominant land cover types. The relationship between land cover change and population density was examined by regression, with rates of land cover change (ALCCR) being the dependent variable and the human population density as independent variable. This analysis was repeated for changes outside IBAs. The land cover change analysis was conducted for the 20-22 year period (1986-2008).

4.3.4. Results

a) Land cover change within IBA correlated with human population density

Population density at the sites varied widely between 9 and 2,701persons per sq.km with the latter being associated with sites in either urbanised landscapes where population density was higher than in non-urbanised areas (Anova: $F_{6, 65}$: 3.146, P=0.0090). Based on Population Reference Bureau (2010), at a national level, population density varies between countries (395 people/sq.km in Rwanda, 306people/sq.km in Burundi, 140people/sq.km in Uganda, 77people/sq.km in Ethiopia, 69people/sq.km in Kenya and 48people/sq.km in Tanzania respectively). There was no significant correlation between the changes in the extent of any of the natural vegetation types and surrounding human population density (Table 19). Even when the human population density was transformed (Log10) to normalise the distribution, the correlation was not significant (Table 20). However, there was perhaps a trend for the extent of tree/shrub crops to increase with increased human population density (Table 19). Based on these results, it therefore suggests that human population density did not contribute much to the model in terms of determining the observed levels and rates of land cover change on these IBAs.

Table 19: Rates of change in all land cover (1986-2008) inside IBAs and human population density

Land cover	Para. Est.	SE	r^2	F	P
All forest	-1.238e-05	1.404e-05	-0.0048	0.777 ($F_{1,46}$)	0.38
Closed forest	1.50e-05	4.650e-05	-0.0279	0.103 ($F_{1,32}$)	0.75
Open forest	-4.149e-05	4.017e-05	0.00143	1.067 ($F_{1,48}$)	0.31
Mosaic	4.614e-05	1.187e-04	-0.0163	0.151 ($F_{1,52}$)	0.70
Shrub land	5.594e-06	7.032e-05	-0.0166	0.0063 ($F_{1,60}$)	0.94
Herbaceous	5.438e-07	2.428e-05	-0.0175	0.0005($F_{1,57}$)	0.98
Tree crop	0.00069	0.0003448	0.1083	4.0361 ($F_{1,24}$)	**0.056**
Arable	1.197e-05	4.947e-05	-0.0171	0.0586 ($F_{1,55}$)	0.81
Water	-9.242e-07	2.220e-05	-0.0333	0.0017 ($F_{1,30}$)	0.97
Flooded veg	-4.193e-05	3.119e-04	-0.0847	0.0181 ($F_{1,29}$)	0.89
Urban	0.000156	0.0002314	-0.0847	0.453 ($F_{1,6}$)	0.53
Bare	-0.00011	0.0005798	-0.0457	0.0389 ($F_{1,21}$)	0.85

Table 20: Rates of change in all land cover (1986-2008) inside IBAs and human population density (log transformed)

Land cover	Para. Est.	SE	r^2	F	P
All forest	0.143	1.1354	-0.02139	0.0158($F_{1,46}$)	0.901
Closed forest	-0.023	0.0625	-0.0269	0.136 ($F_{1,32}$)	0.714
Open forest	-0.223	0.3857	-0.0138	0.3349 ($F_{1,48}$)	0.566
Mosaic	0.214	0.1863	0.0061	1.324 ($F_{1,52}$)	0.255
Shrub land	-0.052	0.2559	-0.0160	0.0407($F_{1,60}$)	0.841
Herbaceous	0.035	0.5865	-0.0175	0.0037($F_{1,57}$)	0.952
Tree crop	0.351	0.2358	0.0463	2.213 ($F_{1,24}$)	0.150
Arable	0.187	0.2972	-0.0109	0.3948 ($F_{1,55}$)	0.532
Water	-0.041	1.2658	-0.0333	0.0011 ($F_{1,30}$)	0.974
Flooded veg	-0.1106	0.2705	-0.0286	0.1671 ($F_{1,29}$)	0.686
Urban	0.0838	0.5367	-0.1619	0.0244($F_{1,6}$)	0.881
Bare	-0.1572	0.2642	-0.0303	0.3539 ($F_{1,21}$)	0.558

b) Land cover change outside IBAs

Land cover change in areas surrounding IBAs was not related to human population density for any of the land cover types except the extent of surrounding urban land (Table 21). This suggests, as may be expected, that the extent of urban land cover increased proportionately in areas where the human population density was high, with this relationship explaining about 20 % of the variation in urban growth.

Table 21: Rates of land cover change (1986-2008) outside IBAs correlated with human population density

Land cover	Para. Estimate	SE	r^2	F	Pr (>|t|)
Closed forest	-5.347e-05	6.249e-05	-0.005612	0.7321($F_{1,47}$)	0.3965
Open forest	1.757e-05	5.162e-05	-0.01496	0.1158($F_{1,59}$)	0.7349
Mosaic	-1.037e-05	1.224e-05	-0.004221	0.7184($F_{1,66}$)	0.3997
Shrub land	-1.760e-05	1.942e-05	-0.0026	0.8213($F_{1,67}$)	0.3681
Herbaceous	-7.784e-06	1.266e-05	-0.0095	0.3781 ($F_{1,65}$)	0.5408
Tree crop	8.024e-05	7.956e-05	-0.000467	1.017($F_{1,36}$)	0.3199
Arable	2.247e-05	3.345e-05	-0.0083	0.4513($F_{1,66}$)	0.504
Water	2.190e-06	6.251e-05	-0.02436	0.00123($F_{1,41}$)	0.9722
Flooded veg	-5.558e-05	7.476e-05	0.01131	0.5527($F_{1,39}$)	0.4617
Urban	1.810e-04	6.856e-05	0.1926	6.965($F_{1,24}$)	**0.01437**
Bare ground	1.581e-05	1.392e-04	-0.03654	0.0129($F_{1,27}$)	0.9104

4.3.5. Discussion

The rates of land cover change in 1986-2008 within IBAs were not correlated with surrounding human population density, with the one exception of woody crops (e.g. tree/shrub crops or plantations), which increased when surrounded by more high human densities. Outside IBAs, there was no significant relationship either, with the exception of urban extent, which also increased with an increase in human population density. In this study, urban was associated with human settlement and areas extensively modified by humans, but not covered by agriculture (e.g. including built up and non-built up areas, urban vegetated areas, and extraction sites). There is therefore evidence that initial human population density influenced urban expansion, with greater expansion in areas of greater initial density. Otherwise there is no evidence to suggest that initial population density influenced land cover change, both within and outside IBAs of the study sites considered under this study. This disagrees with some previous studies that have shown an inverse correlation between human population density and deforestation (Meyer and Turner, 1992; Cohen, 1997), while in other studies, there was a positive correlation between human population density and extinction or threat to mammals (Kirkland and Ostfeld, 1999) birds (Kerr and Currie, 1995) and plants (Thompson and Jones, 1999) as well as deforestation (Palo, 1994; Rock, 1996). However, the evidence is weaker than often believed (Angelsen and Kaimowitz, 1999) because these studies had weakness in analysis and thus correlation could have been spurious data (McKinney, 2001).

Other reviews (Lambin *et al.,* 2001) concur that neither population nor poverty alone constitute the sole and major underlying causes of land-cover change worldwide. McKinney (2001), however, notes that it is also likely that deleterious impact of human population on biodiversity may occur during the early stages of population increase. However, the results from this study should not be reason to dismiss population density as a non-significant factor in driving deleterious land cover change. Instead this variable needs more investigation, especially the underlying effect of legal protection such that certain sites though surrounded by high density of human populations but with human activity exclusively prevented, such sites may not experience much pressure and hence will not experience significant land cover changes driven by human activities.

Many studies have provided evidence that the impacts on biodiversity of human activities are exacerbated by the tendency of human settlement and areas of high biological value to coincide (Abbitt *et al.*, 2000; Balmford *et al.*, 2001a, 2001b; Araujo, 2003; Luck *et al.,* 2004; Schalermann *et al.,* 2004). The fact that birds (species richness) were most strongly correlated with human population density in Africa (Balmford *et al.*,2001a) than in Europe is an interesting fact and supports the findings from this study that there is no positive correlation between rates of land cover change and human population density. If true this could favour biodiversity conservation in Africa. This may not be true if the observed congruence between patterns of human density and biodiversity is just accidental *(*Araujo, 2003). Schalermann *et al.,* (2005) also concluded that agricultural land use was a better predictor of land cover change than human population density. This supports findings from a study that suggested that even though population density around IBAs was higher than in the rest of the continent as a whole we should not be concerned about this factor (Buchanan *et al*., 2009a).

These results are contrary to previous studies that show that areas with more people have undergone greater habitat conversion (Balmford *et al.,* 2001a). By demonstrating that human population density could also be a large-scale surrogate for disturbance, results from Maurer (1996) and Thompson and Jones (1999) also contradict this study. However, the fact that rates of change in land cover are not strongly correlated with human population density may not entirely mean that this variable (other associated activities) does not have a tremendous impact upon land cover change at IBAs.

Therefore, when interpreting these results, there is need for caution and understanding of the context in which IBAs are located and their level of legal protection. These results may not therefore apply for sites that are surrounded by high human population density yet not enjoying any level of protection. Such sites are subject to vagaries of human activities arising from high population density in the surrounding landscape. Already there are fears that despite the overall increase in protected ecosystems, biodiversity is still in decline, owing to inadequate management of existing sites and gaps in the protection of areas deemed priorities for conservation (UN Statistics Division, 2011). Therefore in view of the challenges facing protected areas, their ability and effectiveness to exclude anthropogenic activities may become compromised in future. As for IBAs, even though a very important global network of key biodiversity sites important for avian conservation and other taxa, only 26% of these critical sites are fully protected (UN Statistics Division, 2011) thus making them easily exposed to human encroachment and other underlying pressures and factors driving land cover change associated with high population density. The results may also not apply to land cover change in IBAs in human dominated landscapes where biodiversity and people are competing for the same space. For example, for a globally threatened highland grassland species, Ndang'ang'a *et al.*,(2002) noted that as human population in the grasslands increases, the mean acreage of land holdings decreases and more grassland is converted to other land uses.

The results suggest that human population density was the least significant factor in influencing land cover change in a consistent manner. Perhaps there are other inherent factors other than human population density to explain the rates and extent of change other than population density, which if they had been investigated could have produced different results. In a study on human population size and threats to birds and mammals, McKinney (2001) found out that the pattern of continental population – threat correlation indicated that per capita human impacts are initially very high and diminish with increasing population size.

In this study, even though the correlation between rates of land cover change and human population density is not positive, other human impacts on biodiversity (habitat and species) that are not necessarily direct but underlying and causing land cover change may be profound. Such impacts which may include but are not limited to overharvesting, introduced species (alien invasive species); trade in wild species could possibly be causing more harm than habitat loss through land cover change. Therefore these other factors could explain the rates of land cover change other than population density. For example McKinney (2001) recognises the complexity of the relationship between human population size and extinction or threat because there are extrinsic causes where the impact per person is a reflection of political, technological, economic and other social factors that determine how much negative impact is produced by an average person. These other types of threats directly related with increased pressure from human activities need testing including social factors (policies, economy). For example, even though there was no positive correlation between population density and land cover change, urbanisation (including human settlement) is an increasing phenomenon and which may have pervasive effects on biodiversity especially in human dominated landscapes. The negative and widespread effect of this phenomenon has been studied widely; particularly focusing on effects of urbanisation (traffic, pollution) on selected flora and fauna (Thompson and Jones, 1999).

Therefore, the lack of association between land cover change and human population density could also imply that interventions through other mitigative ways (e.g. policy, interventions) are having an impact. If this is true therefore, an increase in human population density *per se* may not translate into deleterious land cover change.

Also, the limitations of this study are that there was no control over the sizes of IBAs, which varied greatly. Nevertheless, even though population may be seen as not contributing to the observed land cover change, environmental education and awareness needs to be stepped up. This could lead to increased environmental awareness and better understanding of the impact of human actions on biodiversity and help in changing consumption behaviour and attitudes as well as nurturing environmental consciousness amongst the people, which would ultimately reduce per capita consumption and reduce the human footprint on the environment. Environmental, social and economic policies formulated and implemented should be holistic in nature and aimed at safeguarding food security, less overexploitation and pressure on natural resources, while inculcating ethics and environmental consciousness among the population. Land cover change may not be attributed to population growth alone but also due in part to people's responses to changes in economic opportunities and policies, with biophysical and socio-economic trigger events (Lambin *et al.*, 2003). Also the results from this study may have been masked by other factors including the fact that even though sites may be surrounded by high population density, the preoccupation of the people may not be agriculture or activities that are deleterious to the environment.

Alternatively, surrounding dense populations may in fact have less impact on the site if the sites are either fenced to deter human encroachment or the level of enforcement and management effectiveness is so high that no human activity is significant. Urban sprawl for example affects land cover change through the transformation of urban-rural linkages demonstrated through reduced loss of farms, forests, grasslands, wetlands and related ecosystems. Therefore, stabilisation of human population growth and promotion of the wise use of resources within the urban areas complimented with tailor-made environmental education and awareness programmes are critical issues that need to be addressed if biodiversity in densely populated urbanising areas is to be effectively managed and conserved.

Limitations of this study could also have contributed to the observed results. For example, the crude nature of the population density dataset (GRUMP data), could have contributed to non-significant results when analysing the correlation between land cover change at IBAs and the surrounding population density. Limiting the study land cover change between 1986 and 2008 could also have contributed to the observed results and probably future assessments could consider focusing on longer time frames (e.g. 1972-2008).

Besides the crude nature of the GRUMP data, it also would have been appropriate to consider population along an elevational gradient and determine the combined effect of elevation and population density on the observed rates of land cover change when are considered. This study did not also consider the distribution of the population with respect to other land uses. For example, some studies have shown clustering of rapid land cover change along roads and forest edges, (Chatelain *et al.,* 1996; Lambin and Geist, 2006). These intervening factors are all causing deforestation in tropical regions (Geist and Lambin, 2002).

Additionally, subsequent studies may want to consider the level of environmental awareness amongst the population at sites. People make decisions and take action towards protecting or destroying the environment based on information, attitudes, perceptions, and alternatives and hence awareness raising and environmental education can benefit biodiversity conservation practitioners to change individual and group behaviours around specific environmental issues. If a population is aware of the impacts of their actions on nature, they may adopt sustainable uses of natural resource. Therefore, more assessment should be undertaken integrating other demographic variables as well as using longer time scale data. The same view is expressed by Lambin and Geist (2006).

4.4. Pressures on IBAs (ii): Land cover change on IBAs correlated with agricultural intensity around IBAs

4.4.1. Introduction

Conflicts between biodiversity and development are manifested through many forms of human activities some of which have a very large impact on biodiversity conservation. According to Kleijn *et al.*, (2010), effective conservation strategies depend on the type of relationship between biodiversity and land-use intensity. For example, whereas agricultural intensification provides food security, it is one of the main drivers of biodiversity loss worldwide (Donald *et al.,* 2001; Green *et al.,* 2005). The area of cropland has increased globally between 4.5 and 5 times from 1700 to 1990 (i.e. from an estimated 300–400 million ha in 1700 to 1500–1800 million ha in 1990 and a 50% net increase just in the twentieth century (Lambin *et al.,* 2003). World food demand is expected to more than double by 2050 with a profound effect on biodiversity (Green *et al.,* 2005*)*. Agricultural expansion and intensification has been one of the major drivers of past biodiversity loss and ecosystem degradation globally in the 20th century and the fact that these trends may continue through the 21st century is a major cause for concern (Norris, 2008), especially since expansion may be greatest in areas of high conservation importance (Scharlemann *et al.,* 2005).

All major habitats for species have not been spared the vagaries of agricultural intensification. For example, the dominant force in forest loss is the ever growing demand for farmland such that through this economic activity, humans have converted 36% of the Earth's land surface area to agriculture at the expense of natural habitats (Morris, 1995) making agricultural expansion one of the key direct causes of forest ecosystem change and biodiversity loss (Slingenberg *et al.*, 2009). Agricultural intensity has been associated with the decline in farmland birds (Donald et. al., 2001) as well as a decrease in bird assemblages (Luck and Daily, 2003) and farming is the major current and likely future threat to globally Threatened and Near-Threatened bird species, especially in developing countries where 1,726 out of all 1,923 birds in these threat categories are found (Green *et al*., 2005).

Expanding and intensifying agriculture often results in encroachment, modification and conversion of natural habitats. This intensification can also lead to a decline in farmland biodiversity as a result of the emergence of agricultural systems and practices that are not friendly to biodiversity conservation (Turner *et al.,* 1990). Because of the expansion and intensification of agriculture, key biodiversity sites end up being surrounded by a very harsh agricultural matrix varying from low to medium to high and characterised by either mosaics of subsistence to monocultures and commercial/plantations which potentially affect ecological processes (e.g. movements of organisms, fire, other disturbance regimes and water availability) within the key biodiversity sites or protected areas (DeFries *et al.,* 2010).

Agricultural expansion and intensification is also associated with other factors such as the increasing population density and the proportionate demand for settlement and food production. For example, the expansion of human populations and commodity markets causes the conversion of natural vegetation into farmland and its intensifying use, which in turn renders habitats for wild flora and fauna smaller, more fragmented, and less diverse. Agricultural intensification in a poor and densely populated developing region in southern Uganda affected the abundance and species richness of woody vegetation and birds (Bolwig *et al.*, 2005). DeFries *et al.*, (2010), suggest that land use and a number of detrimental anthropogenic forces outside protected area boundaries potentially affect ecological processes within protected areas. With all these changes taking place in and around key biodiversity sites, IBAs have not been spared, therefore making the need to quantify the impact of anthropogenic forces around them a priority. In this particular study, the intensity of agricultural activities outside IBAs is used to determine if it is driving land cover change both within and outside the IBAs. Higher-scale studies on the effects of land-use change on biodiversity have focused mainly on 'major land-use types' with little attention paid to the intensity of land use (e.g. Sala *et al.*, 2000). This study addresses this gap by focusing on agricultural intensity as a land use factor driving land cover change at key biodiversity sites such as IBAs.

4.4.2. Aims

This section examines whether land cover change in IBAs is related to the extent of agriculture in the surrounding area. Specifically, by correlating rates of land cover change with extent of agriculture in the areas surrounding the IBAs, this study assessed whether these rates increased when agricultural extent was higher. The following hypotheses were tested:

a) *The level of agricultural extent is correlated with the rate of land cover change at IBAs*

b) *Land cover change is greater at IBAs sorrounded by high agricultral extent*

4.4.3. Methods and Analysis

The extent of, and change in six broad land cover classes (artificial landscape, forest, grassland, natural-agricultural mosaic, shrub land and wetlands) on IBAs were compared to the extent of agriculture within a 20 km buffer around each IBA. Extent of agriculture in the surrounding matrix was derived by determining the proportion of land cover in and outside IBAs that was arable during the initial period (1986). The extent of agriculture in the surrounding matrix was an independent variable while rates of change in land cover types were dependent variables. The rates of change in the respective land cover categories both within and outside IBAs in this analysis were derived for one time period (1986-2008) as a precaution against multiple testing issues.

Regression analyses between dependent and independent variables were done in R statistical software. Generalised regression models were used to investigate the relationships in terms of the level of significance of the extent of agriculture in the surrounding matrix as a confounding variable on rates of land cover change at the IBAs. In the first analysis, rates of change in all natural vegetation, forests, dense forest, open forest, natural-agricultural mosaic, shrub land, herbaceous, tree/shrub crop, arable, water, flooded vegetation, urban and bare was determined. The results were tabulated showing the levels of significance for each regression analysis.

4.4.4. Results

a) Initial extent of agriculture and Land Cover change Inside the IBAs

Rates of all forest cover change (1986-2008) inside IBAs were negatively correlated with the extent of agriculture surrounding IBAs (Table 22; Figure 19), suggesting that the extent of forest decreased more on IBAs which were surrounded by a greater cover of agriculture. The same significant relationship was observed when open forest was treated alone, and also for shrub land and herbaceous cover. Scatter plots showing the relationships between rates of change in respective land cover types and agricultural extent are shown in Figures 20-23 below.

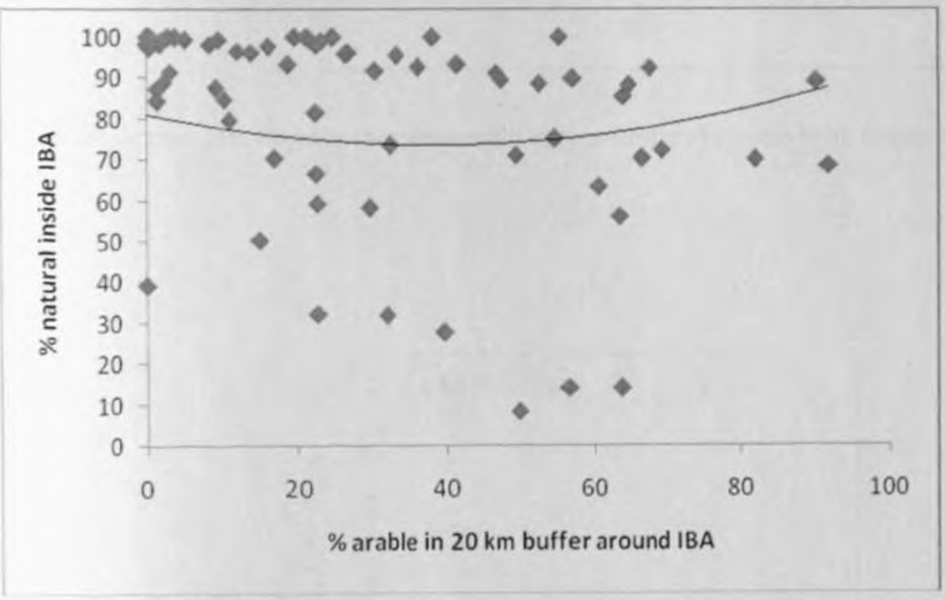

Figure 19: Relationship between rates of change in forest land cover inside IBAs and extent of agriculture surrounding the IBAs. (R2= 0.1722, F (3, 39) = 9.321, p= 0.004067)

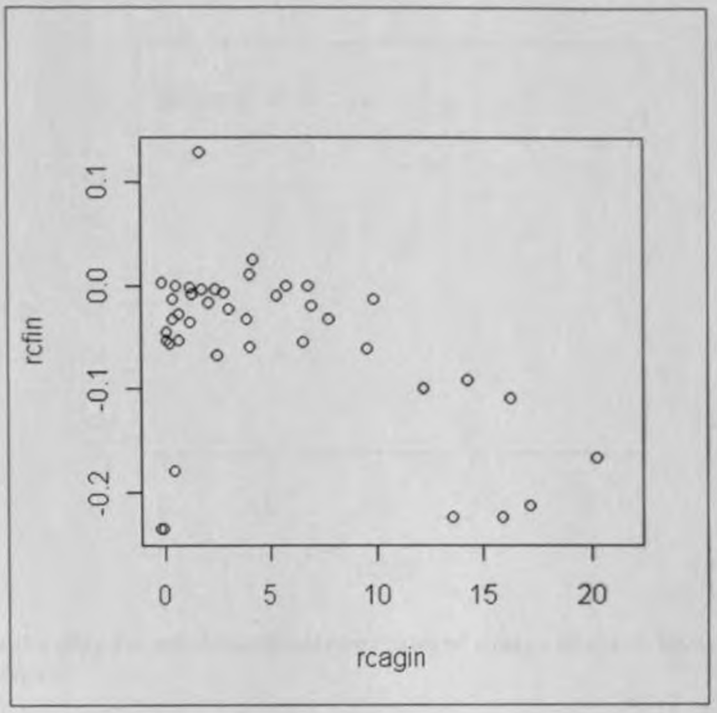

Figure 20: Scatter plot showing the relationship between rates of change in all forests and agricultural extent

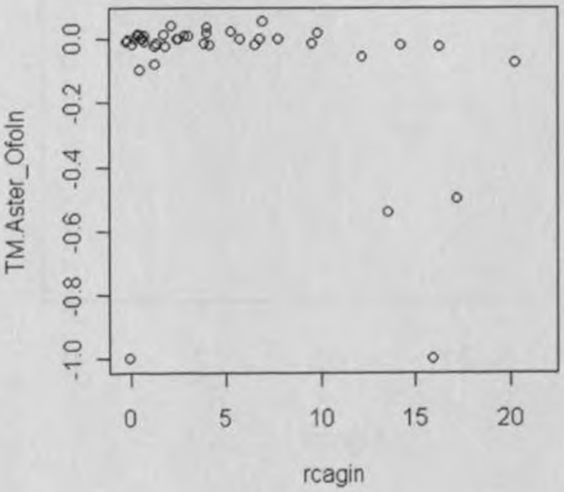

Figure 21: Scatter plot showing the relationship between rates of change in open forests (1986-2008) and extent of agriculture outside IBAs (TM.Aster_OfoIn= change in open forest, rcagin=agricultural intensity)

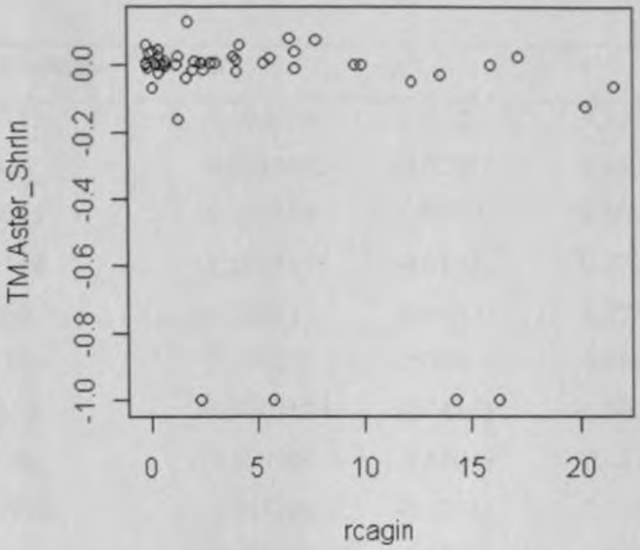

Figure 22: Scatter plot showing the relationship between rates of change in shrub land (1986-2008) and extent of agriculture outside IBAs

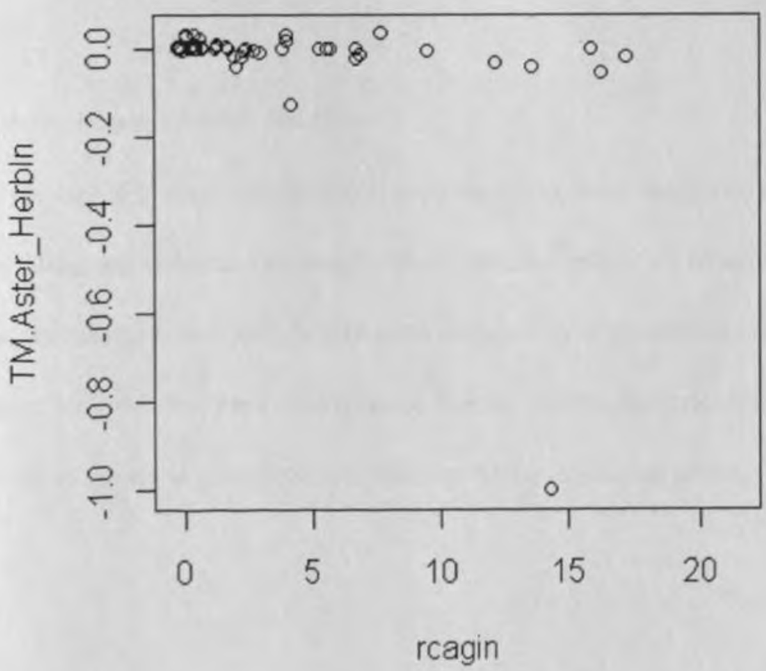

Figure 23: Scatter plot showing the relationship between rates of change in herbaceous (1986-2008) and extent of agriculture outside IBAs (TM.Aster_HerbIn represents change in herbaceous inside IBA and rcagin represents agricultural intensity)

Table 22: Rates of land cover change (1986-2008) inside IBAs correlated with the extent of agriculture surrounding IBAs

Land cover	Para. Estimate	SE	r^2	F	Pr (>|t|)
All forests	-0.006331	0.002074	0.1722	9.321($F_{1,38}$)	**0.004**
Closed forest	0.005356	0.008168	-0.02157	0.43($F_{1,29}$)	0.518
Open forest	-0.013789	0.006366	0.08262	4.692 ($F_{1,40}$)	**0.036**
Mosaic	-0.002349	0.008315	-0.01955	0.07977 ($F_{1,47}$)	0.778
Shrub land	-0.01356	0.00672	0.06011	4.07 ($F_{1,47}$)	**0.049**
Herbaceous	-0.011167	0.004261	0.1154	6.868 ($F_{1,44}$)	**0.012**
Tree crop	0.000565	0.0108174	-0.04335	0.002724 ($F_{1,23}$)	0.959
Arable	0.002706	0.003901	-0.01049	0.4811($F_{1,49}$)	0.491
Water	0.0005258	0.0042049	-0.03935	0.01564 ($F_{1,25}$)	0.902
Flooded veg	0.00364	0.02344	-0.04239	0.02412($F_{1,23}$)	0.878
Urban	-0.02400	0.04745	-0.1416	0.2559 ($F_{1,5}$)	0.635
Bare ground	0.004037	0.029097	-0.06122	0.01925 ($F_{1,16}$)	0.891

b) Land Cover change Outside the IBAs

Rates of all forest cover change (1986-2008) outside IBAs were negatively correlated with the extent of surrounding agriculture. The results show that the extent of natural-agricultural mosaic and herbaceous decreased more outside IBAs surrounded by a greater cover of agriculture. This therefore suggests but does not provide evidence that the extent of agriculture has a strong effect on the rates of change in natural agricultural mosaic, herbaceous and urban.

Table 23: Rates of land cover change (1986-2008) outside IBAs correlated with the extent of agriculture surrounding IBAs

Land cover	Para. Estimate	SE	r^2	F	Pr (>|t|)
Closed forest	-0.004414	0.00982	0.04502	0.2022 ($F_{1,36}$)	0.6556
Open forest	-0.013392	0.00752	0.04502	3.169 ($F_{1,45}$)	0.08182
Mosaic	-0.001691	0.00083	0.05666	4.183(**$F_{1,52}$**)	**0.0459**
Shrub land	-0.002759	0.00284	-0.001082	0.9427($F_{1,52}$)	0.3361
Herbaceous	-0.001571	0.00049	0.1552	10.18($F_{1,49}$)	0.**0025**
Tree crop	0.006837	0.00865	-0.01151	0.6243 ($F_{1,32}$)	0.4353
Arable	0.007051	0.004812	0.02158	2.147 ($F_{1,51}$)	0.1490
Water	-0.02508	0.01445	0.05582	3.01 ($F_{1,51}$)	0.09208
Flooded veg	-0.01507	0.01127	0.02261	1.786 ($F_{1,33}$)	0.1905
Urban	0.03598	0.01663	0.1624	4.683($F_{1,18}$)	**0.04416**
Bare ground	-0.008787	0.018248	-0.03455	0.2318($F_{1,22}$)	0.6349

4.4.5. Discussion

Agricultural intensification permitted the doubling of the world's food production from 1961 to 1996 with only a 10% increase in arable land globally (Tilman, 1999). For example, the area of cropland has increased globally between 4.5 and 5 times from 1700 to 1990 (i.e. from an estimated 300–400 million ha in 1700 to 1500–1800 million ha in 1990 and a 50% net increase just in the twentieth century (Meyer and Turner II, 1992); Lambin *et al.,* 2003). Comparing the increase in the area under agriculture at the continental level within a period of 20 years (1980-2000), agricultural area increased by 3% from the initial 1,102, 575, 000ha to 1,136,660,000ha whereas arable land also increased by 15% during the same period from 158,354,000ha to 181,409,000ha (UNEP-DEWA, 2008). In a study by Brink and Eva (2009), agriculture increased by 57% during a 25 year period in the sub Saharan Africa. As a result, expansion and intensification of agriculture is now recognized as one of the most significant human alterations of the global environment (Matson *et al.*, 1997).Yet Tilman (1999) reports that recent intensification of agriculture, and the prospects of future intensification, will have major detrimental impacts on the nonagricultural terrestrial and aquatic ecosystems of the world. This is partly because agricultural intensity is known to replace natural vegetation (Matson *et al.*, 1997).

This study shows that land cover change within IBAs generally increased with greater agricultural intensity in the surrounding landscape. The main habitats within IBAs that are affected by the extent of agriculture in the surrounding area include all forest habitats, open forests, shrub land and grassland IBAs. These findings confirm initial conclusions by Fishpool and Evans (2001) and Buchanan *et al.,* (2009a) that agricultural expansion and deforestation are the most prevalent threats affecting IBAs. With the results of the preceding chapter it supports Scharlemann *et al.,* (2005) that land cover (interpreted here as land use) is more important than population in determining rates of natural habitat loss. This means that during the period (1986-2008), an increase in agriculture led to increase in the decline in land cover of these habitats at IBAs. This result also concurs with Scharlemann *et al.,* (2004) who found that agriculture increases in EBAs according to models and that the proportion of land in agricultural use is currently greater in EBAs than in the rest of the world.

Previous studies have also provided an overview of interactions between key biodiversity sites, protected areas and the adjacent areas (Buchanan *et al*., 2009a; DeFries *et al.,* 2010). Tilman *et al.,* (2001), predicts that human population increase will lead to continued clearance of land for agriculture as well as increased agricultural intensification with serious ramifications in biodiversity loss. Agricultural development is a leading cause of habitat destruction that increasingly threatens global biodiversity (Gorenflo and Brandon, 2005) including being a decisive factor in species extinction (Vitousek *et al.,* 1997).Therefore considering the results from this study and the implications of this as demonstrated by other studies, this increase and intensification in agriculture should be a matter for concern for conservation practitioners.

However, other related studies have found conflicting results. According to (Tiffen *et al.,* 1994) agricultural intensification did not lead to long-term environmental degradation, but to environmental improvement including increased investments in land and improved conservation of natural resources. However, Tiffen *et al.,* (1994) have been criticized for not addressing the biodiversity issue (Rocheleau, 1995).

In general, as in this case the loss of forests, shrub lands and herbaceous land cover within IBAs due to modification and conversion to agriculture and increased agricultural intensity will have a strong impact on the survival of species that are restricted to this type of habitat (e.g. forest dependent birds, shrub land dependent birds and grassland dependent birds) as a result of the change in ecosystem structure and functioning. However, the lack of significant results for the other land cover types may be due to the ease with which vegetation change can be identified. Thus, we may expect to detect changes in forest easier than other land cover types, but the lack of relationship with dense forest, and the detection of a relationship with grasslands, suggests this is not the case. This could also be attributed to sample size with a possibility that the same size of other land cover types may have been smaller. This is partly because transformation of natural vegetation is more pronounced and in most cases these changes affect all natural land cover types including dense forest, open forest, shrub, herbaceous and flooded vegetation. There is a cumulative land cover change across all these land cover types at the IBA covered in this study. A similar but weaker pattern is observed when considering the relationship between agricultural intensity and land cover change from dense forest land cover category to other land cover types or agriculture.

Even though the rates of land cover change in the dense forest category is prevalent at certain sites, this significant change may be masked as a result of lumping together many sites some of which are associated with land cover changes in other land cover categories other than dense forest. However, there may be other factors at play that could have masked the influence of agricultural intensity on land cover change. For example, the impact of agricultural intensity may have varied significantly from site to site as a result of other underlying factors such as protection status, difficulty of the terrain, remoteness, quality of the soil in the surrounding matrix, management regimes and effectiveness of the protected areas as well as how effective site-based conservation interventions are, particularly those that involve working with communities around the sites. There may be cases where the protection could have been so effective at a site that there was no encroachment regardless of the agricultural intensity around IBAs.

Nevertheless, the extent of surrounding agriculture has an important effect on land cover change in IBAs. In terms of the threats on various land cover types affected by the extent of agriculture in this study, previous studies (e.g. MacDonald *et al.*, 1993, Reyers *et al.*, 2001) show that grasslands are threatened in many locations in Africa. Agronomy and animal husbandry are major land uses in grasslands (USAID, 2005). A study by Ndang'ang'a *et al.,* (2002) confirms this by providing evidence that agricultural encroachment was the biggest threat to a highland grassland IBA.

The land cover and land use dynamics within the matrix surrounding IBAs has an enormous impact on the biodiversity within the agricultural landscape. The study reports that there is a significant correlation between agricultural intensity and the decline in natural-agricultural mosaic habitats, shrub land and herbaceous cover outside IBAs as opposed to other land cover types. Outside IBAs, as agricultural intensity increases, this not only puts great pressure on the site but equally, small and isolated patches of forests and woodland vegetation interspersed with agriculture disappear once they are converted to agriculture or alternative land uses. The same applies to shrub land and herbaceous cover which gets modified and converted to agriculture as the demand for agricultural land and settlement. However, the negative implication of this is reduced land cover for biodiversity, extirpation of corridors which may have been appearing as linear patches of natural habitat that link or connect fragmented areas, allowing species movement and ecological processes to continue providing shelter and facilitating movement between natural vegetation pockets. The elimination of corridors and stepping stones may also expose sites and species to the vagaries of edge effects (Lovejoy *et al.,* 1986).

The correlations suggest that land use around key biodiversity areas could be affecting land cover within it, in reducing the effective size of habitat and concomitant extirpation of species, reduced species richness (Terborch, 1990; Vester *et al.,* 2007). Regarding the impact of agricultural expansion at IBAs, Buchanan *et al.,* (2009a), concludes that agricultural land cover was the fourth largest habitat in IBAs and hence an indicator that there has been considerable agricultural development within IBAs. Species will therefore be forced to adapt to the new modified habitats or mitigative measures will have to be undertaken to restore these areas if species that cannot be supported by these natural-agricultural mosaics are to survive in the long-term.

In Eastern Africa where this study was focused, previous studies show that cropland expanded by 200% between 1900 and 1990 (Klein, 2001). The fact that deforestation or in general land cover change is driven by expansion of smallholder agriculture for subsistence purposes and fuel wood extraction (Lambin *et al.,* 2003) and compounded by a growing high population pressure particularly in Kenya and Uganda (UNEP-DEWA WRCF, 2001) should be a major cause for alarm. This threat needs to be addressed urgently because studies conducted elsewhere have shown that populations of species have declined. For example, in Europe, agricultural intensification had continent-wide consequences for individual bird species (Green, 1996; Wilson *et al.,* 1997) and bird communities (Campbell *et al.,* 1997; Newton, 1998). However, agricultural landscape within the matrix surrounding key biodiversity sites (e.g. IBAs) can if well planned also contribute to biodiversity conservation through sustainable, environmentally friendly farming systems (Kaihura and Stocking, 2003; USAID, 2005).

The agricultural landscape may itself be vital in the management of biodiversity within human dominated and isolated natural habitat landscapes (Eshiamwata *et al.,* 2006) and therefore environmental friendly farming systems have a large bearing on biodiversity. Therefore leaving intact corridors, forest patches or gallery forests in the surrounding matrix will support connectivity and movement of species between various fragments or patches of natural habitats. This can be complimented with agri-environment schemes and farming systems that are sustainable and biodiversity friendly even though the lessons learnt from other regions (e.g. Europe) where effectiveness of agri-environmental schemes have been interrogated (Kleijn *et al.*, 2006) show that they are yet to provide detectable biodiversity benefits (Davey et al., 2010).

These results confirm evidence by Hendrickx *et al.,* (2007) that agricultural intensification poses a serious threat to biodiversity as a consequence of increased land-use intensity, decreased landscape heterogeneity and reduced habitat diversity resulting from deleterious land cover change. Africa has recorded increased area under cultivation while per capita food production has declined (World Resources Institute, 1994). In view of the looming food insecurity (e.g. famine and starvation) worldwide and the conversion of land to biofuels, the demand for agricultural land is set to increase substantially hence putting great pressure on natural habitats. Strategies have to be developed to ensure that this does not continue to contribute to detrimental land cover change that would affect species survival and the ability of biodiversity to provide ecosystem services. For example, a study by Dewi *et al.,* (2012) found out that agroforestry practices can maintain or increase the degree of integration of forest in the multifunctional landscape for biodiversity and local people's livelihoods. Including buffer areas outside protected areas may not only reduce pressure from these key biodiversity sites but has the potential to also mitigate the displacement of habitat loss from these sites to the surrounding matrix.

4.5. Are Site-based conservation approaches effective? : (i) Land cover change at IBAs correlated with Protected Area status

4.5.1. Introduction

The most important direct drivers of biodiversity loss and change in ecosystem services are habitat change from habitat transformation on conversion to agriculture, over-exploitation, and biotic exchange from alien species, pollution and global climate change (Millennium Ecosystem Assessment, 2005). Conserving biodiversity through a site-based approach is one of the key components of any strategy to reduce biodiversity loss (Boyd *et al.*, 2008) and therefore to address this complex array of factors, protected areas are being created and designated worldwide with the objective to protect species and ecosystems. Therefore, protected areas are often the primary defence against species extinctions and habitat loss (Pimm *et al.*, 2001, Chape *et al.*, 2005). A Protected Area is, according to Dudley (2008), "A clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values". The importance of large-scale planning for conservation is gaining increasing recognition and countries are now investing substantial efforts in developing a comprehensive system of protected areas network.

Protected area coverage indicator measures policy response to biodiversity loss because an increase in protected area coverage indicates increased efforts by governments and civil society to protect land and sea areas (UNEP-WCMC, 2010). The increase in the protected area network is not only phenomenal but is also a demonstration of attempts and commitments by State Parties to pursue global conservation targets who have acknowledged the importance of protected area networks (Dudley *et al.*, 2005). The significance attached to protected areas by governments is illustrated by their inclusion as a target in the CBD and MDG (Secretariat of the Convention of Biological Diversity, 2008; Pistorius *et al.*, 2008).

For example, in 1992, State Parties to CBD set a target of 12% of global land surface to be covered by protected areas by 2010 (IUCN, 1993). By 2010, this target had been surpassed since over 150,000 protected sites covered 12.7 per cent of the world's land area and 7.2 per cent of its coastal waters (extending out to 12 nautical miles) was part of this comprehensive network of protected areas. As part of a strategic goal to improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity, a new target (e.g. Target 11[3]) of at least 17% protected area coverage by 2020 was set in during the CBD Conference of Parties meeting in 2010.

The gaps in the extent of marine protected areas coverage notwithstanding, the overall increase in protected area is a good indicator of how biodiversity conservation through site-based action is becoming popular worldwide. However, the caveat is that biodiversity is still declining due to the existing mismatch between protection and improved management effectiveness. This is according to UN Statistics Division (2011) as a result of owing to inadequate management of existing sites and gaps in the protection of areas deemed priorities for conservation (e.g. IBAs and Alliance for Zero Extinction Sites). Even though they may not be sited in the best areas for biodiversity conservation (e.g. Rodrigues *et al.,* 2003; Beresford *et al.,* 2010), Protected Areas (PAs) are a fundamental part of biodiversity conservation and have been recognised at local, national, regional and global levels (Chape *et al.*, 2003) with demonstrable varying cases of

[3] Target 11: By 2020, at least 17 per cent of terrestrial and inland water areas and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscape and seascape.

success in achieving their objectives (i.e. reduced deforestation, Sánchez-Azofeifa 1999; Nepstad *et al.*, 2006; Nagendra, 2008; Brooks *et al.*, 2009). The increase in the number and coverage of protected areas worldwide represents the past century's most notable conservation success even though many protected areas are ineffectively managed (Ervin, 2003a) owing to many challenges including inadequate capacity and funding and hence this success could be disputed. This is against the backdrop of protected areas absorbing a high proportion of global conservation investment (Chape *et al.*, 2005, Chape *et al.*, 2008). In a study by Leverington *et al.*, (2010), aspects of management relating to the establishment of protected areas were relatively strong but the management planning was very weak. Even though they may not be sited in the best areas for biodiversity conservation (e.g. Rodrigues *et al.*, 2003, Beresford *et al.*,2010), Protected Areas (PAs) are a fundamental part of biodiversity conservation and have been recognised at local, national, regional and global levels (Chape *et al.*, 2003) with demonstrable varying cases of success in achieving their objectives (i.e. reduced deforestation, Sánchez-Azofeifa, 1999; Nepstad *et al.*, 2006; Nagendra, 2008; Brooks *et al.*, 2009).

However, even though efforts are being made to increase the coverage of protected areas, the need to assess the effectiveness of the protected areas already available is critical. There exists considerable uncertainty and controversy over the impacts and effectiveness of protected areas (e.g. Nelson and Chomitz, 2009) and dearth of rigorous and very few well- designed evaluations of the impact of specific interventions (Chomitz, 2007). This justifies the need to evaluate the impact of these areas in delivering conservation targets and reducing biodiversity loss. Protected areas have been designated to conserve biodiversity but evaluating the level of success in achieving this objective is not only challenging regardless of the level at which it is assessed (e.g. detailed park study, regionally, Nepstad *et al.*, 2006) or globally (Bruner *et al.*, 2001).

The area of evaluating effectiveness of protected areas has received considerable attention in recent years (Bhagwat *et al.*, 2001; Figueroa and Sanchez-Cordero, 2008). However, according to Dudley (2008), one area of study that has received even more attention is the assessment of the management effectiveness (e.g. how well protected areas are being managed). Leverington *et al.*, (2010) identifies 9, 250 specific Protected Area Management Effectiveness studies that have been undertaken from 6,720 protected areas, derived from 54 different methodologies. These studies show that the effectiveness also varies from site to site, country to country and region to region as a result of other intervening variables.

For example, Liu *et al.*, (2001) noted that though considered a haven for species and other forms of biodiversity, many protected areas are also subject to the vagaries of threats including human encroachment. Some have used remote sensing to assess the degree to which protected areas in the tropics have become isolated from surrounding ecosystems through loss of habitat in their surrounding matrix (DeFries *et al.*, 2010). In this study they stressed the need for scientifically-based regional land use planning to balance human needs and conservation goals in the larger landscape and improved understanding of the ecological processes affected by land use in the matrix surrounding protected areas. Encroachment is usually manifested through deleterious land cover change and while some studies have indicated protected areas may be effective in reducing vegetation change (Ervin, 2003b), others have shown that designation does not necessarily reduce habitat loss (e.g. Buchanan *et al.*, 2008).

Many of these studies have focused on forest loss particularly in the tropical forest of America, meaning that the pattern for other regions and land cover types remains largely unstudied. The need to understand land cover change and losses of habitat in protected areas has resulted in monitoring of protected areas using remote sensing (Sánchez-Azofeifa *et al.*, 1999; 2004; Kinnaird *et al.*, 2003; Linkie *et al.*, 2004). These rapid changes require spatially explicit monitoring for which remote sensing becomes very useful because of its cost effectiveness (Trigg *et al.*, 2006). Further, even where land cover change studies have been conducted, most of these studies have focused on key biodiversity sites or protected areas only yet land cover and land use change is occurring in lands surrounding protected areas around the world particularly the tropics (Joppa *et al.*, 2009b). Most studies of land cover change at protected areas have mainly focussed on changes within the protected area boundaries, the dynamics in the surrounding matrix notwithstanding. This is despite land cover change outside key biodiversity sites having a severe impact on these sites. For example, the intensity of land use around key biodiversity sites will not only affect biodiversity in the surrounding landscape but makes even the biodiversity within the site itself susceptible such that pressures outside may encroach and impact on the integrity of the site.

In a study by DeFries *et al.*, (2010), land use change and other anthropogenic forces outside key biodiversity site boundaries potentially affect ecological processes within a site. In studies evaluating the effectiveness of protected areas taking into account "leakage", which is the degree to which protection of one forest plot (or area) merely displaces conversion to another, unprotected plot or area (Nelson and Chomitz, 2009) is critical. For example, direct comparison of rates of land cover change within PAs and their immediately surrounding areas may be biased by the effects of 'leakage' (Ewers and Rodrigues, 2008), whereby reducing pressure on natural resources within a PA leads to a concomitant increase in pressure immediately outside it.

This study therefore evaluates the effectiveness of protection as well as the dynamics within the surrounding matrix to determine if protection contributes to leakage, therefore conferring a profound effect on the site. This therefore provides a justification to analyse land cover change around key biodiversity sites such as IBAs and protected areas.

4.5.2. Aims

This section aims to determine whether protection status is effective in achieving its core objectives of preventing and stemming the current rates of land cover change and conserving biodiversity. The relationship between rates of land cover change and the protection status of IBAs was examined. It assesses how various land cover types at protected and non-protected IBAs have changed and whether these deleterious changes are more pronounced at unprotected sites than protected ones. One hypothesis is tested in relation to land cover change and protection area status:

The rate of land cover change is higher at non-protected areas than protected

4.5.3. Methods and Analysis

The Protected Area status for each of the focal IBAs was extracted from UNEP World Database on Protected Areas (IUCN and UNEP, 2008); including the year it was established and thus numbers of years since its designation as a protected area. Where this chronological information was not available from the UNEP database, additional information was gleaned from literature, websites as well as from the BirdLife International IBA database. The efficacy of protection at preventing land cover change was assessed by quantifying the rates of change, and then the total extent of change between 1986-88 and 2006-08 as well as the percentage of areas transformed during that period. A total of 71 sites were used in this analysis out of which 41 were protected areas falling in any of the IUCN protected area categories, 2 were partially protected areas and 28 were not protected areas.

The effectiveness of legal protection in reducing land cover change in areas of recognised conservation importance across Africa was quantified using visual interpretation of satellite images. For example, Land cover change in each land cover category for each site was assessed using the Land Cover Change – Graphical User Interface (Chapter 3) tool. The analysis involved land cover change within and outside the focal protected areas and non-protected areas and across all major habitat types.

Annual land cover change rates (ALCCR) were evaluated as $ALCCR = (S1/S2)^{1/N}-1$

Where S1 = Final % Land cover, S2 = Initial % Land cover and N= Time lag in years

Test for normality of data was conducted in SPSS. Two independent samples T-Tests were conducted to compare whether rates of land cover change on protected and unprotected IBAs differed. In this case, comparison of data was from two independent sets of data. Land cover change at IBAs was correlated with protected area status. General Linear Models were used for controlling local rates of change (e.g. rates of change in the surrounding area) to investigate if protection has an effect.

4.5.4. Results

Tests of normality of data resulted in the normal Q-Q probability plots for rates of change (1986-2008) for respective land cover types as presented in Figure 24. Most of the points exhibited linearity as the points cluster around a straight line. Figures 24-33 represents box plots for change inside and outside unprotected and protected areas.

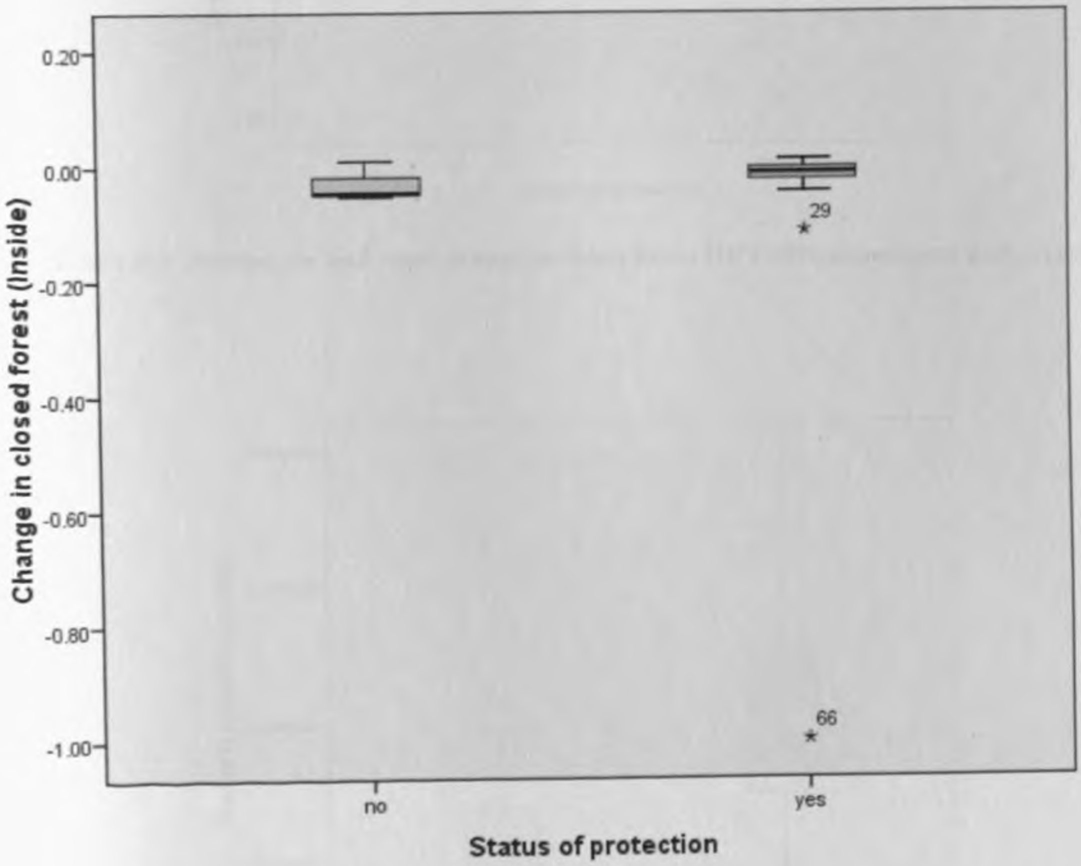

Figure 24: Box plot showing the land cover change in closed forest INSIDE unprotected and protected IBA sites

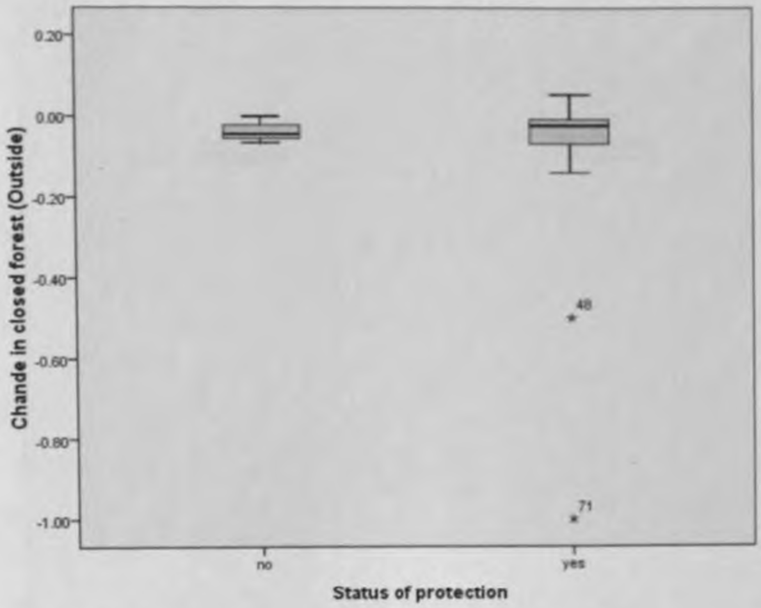

Figure 25: Box plot showing the land cover change in closed forest OUTSIDE unprotected and protected IBA sites

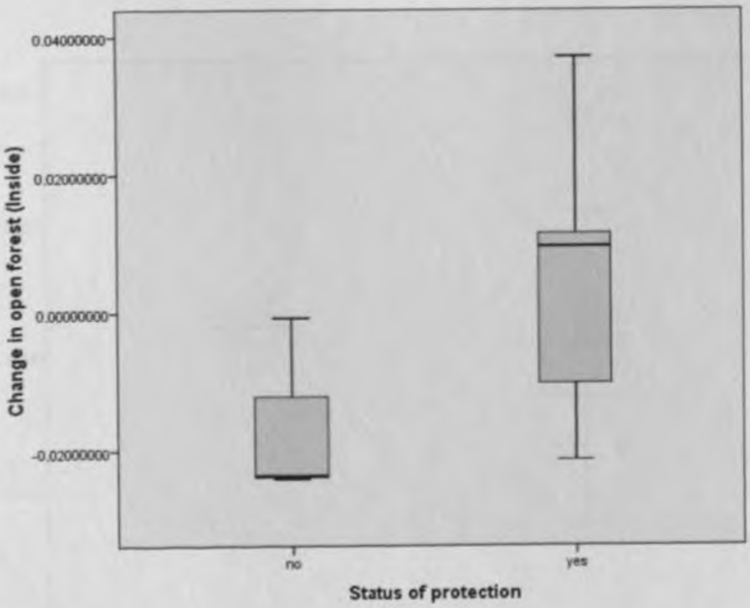

Figure 26: Box plot showing the land cover change in open forest INSIDE unprotected and protected IBA sites

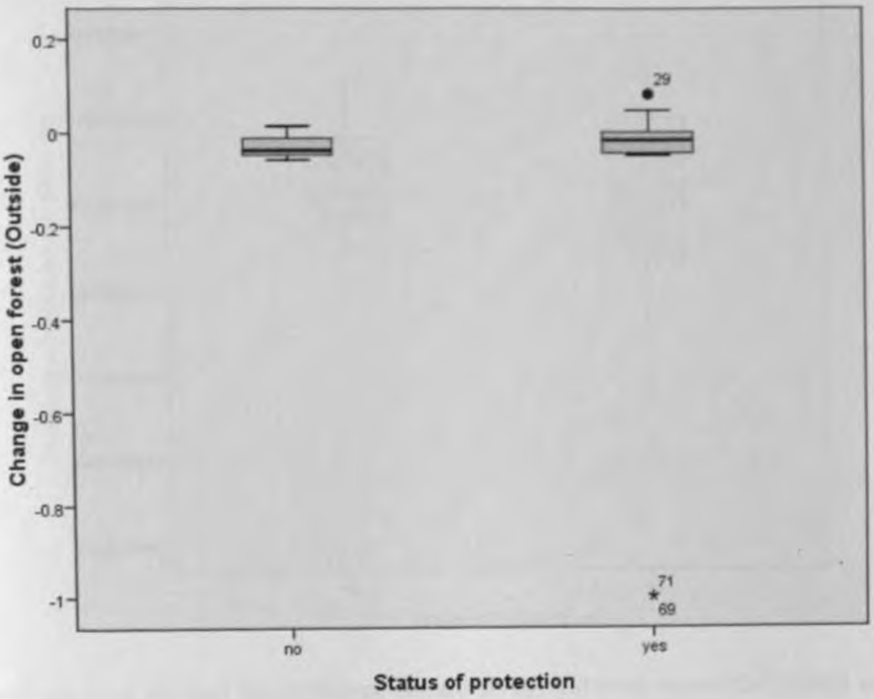

Figure 27: Box plot showing the land cover change in open forest OUTSIDE unprotected and protected IBA sites

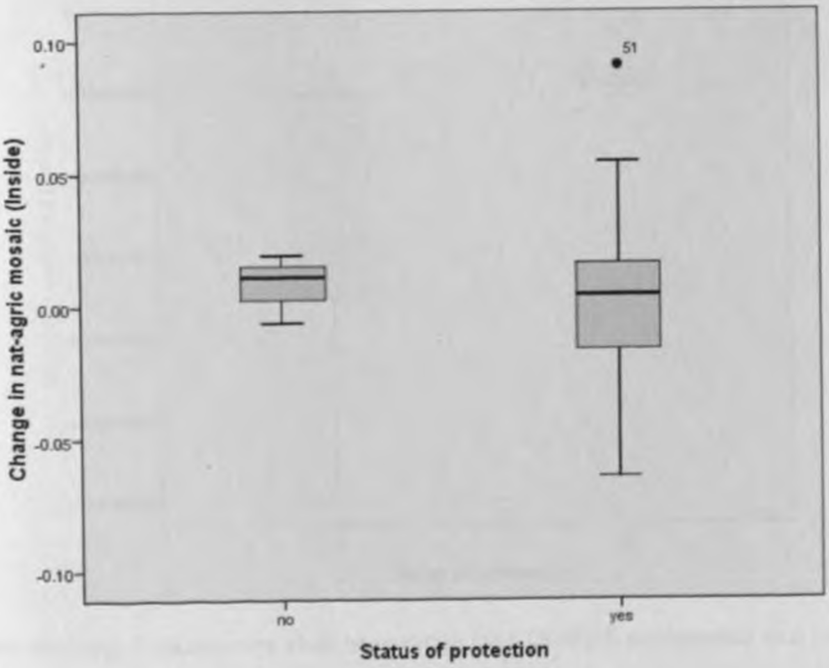

Figure 28: Box plot showing the land cover change in natural-agricultural mosaic INSIDE unprotected and protected IBA sites

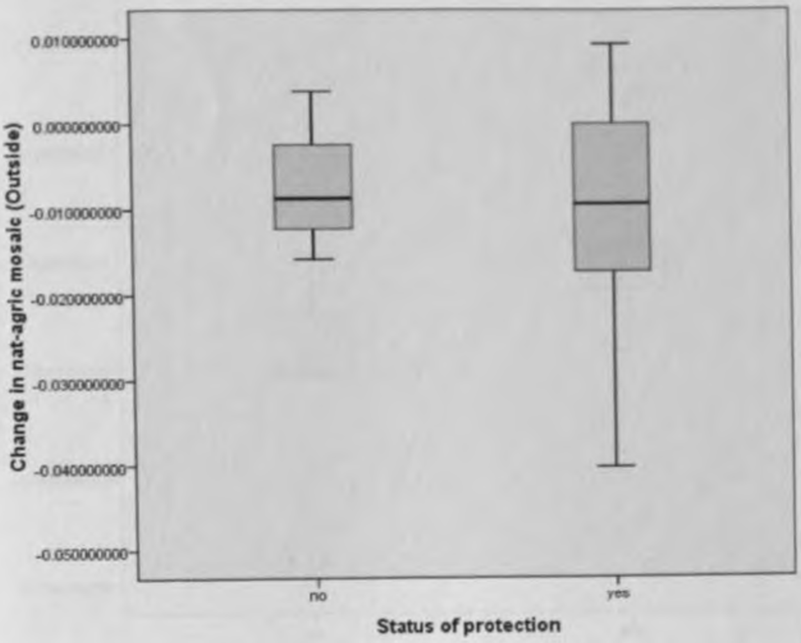

Figure 29: Box plot showing the land cover change in natural-agricultural mosaic OUTSIDE unprotected and protected IBA sites

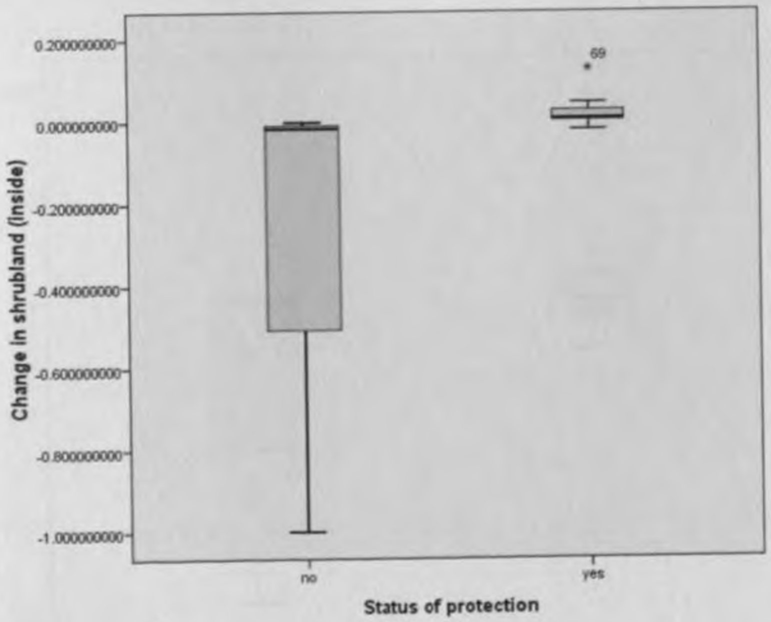

Figure 30: Box plot showing the land cover change in shrub land INSIDE unprotected and protected IBA sites

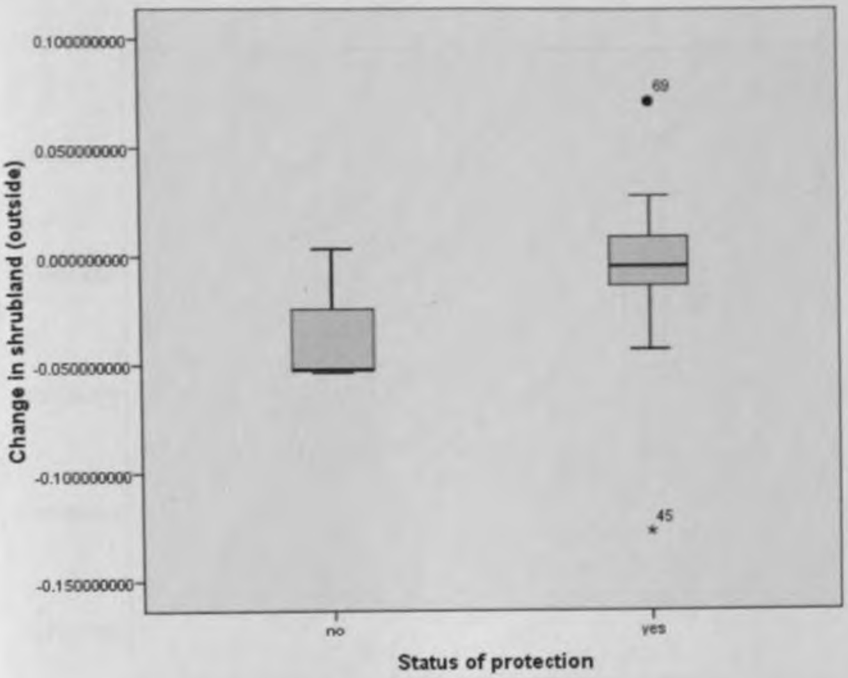

Figure 31: Box plot showing the land cover change in shrub land OUTSIDE unprotected and protected IBA sites

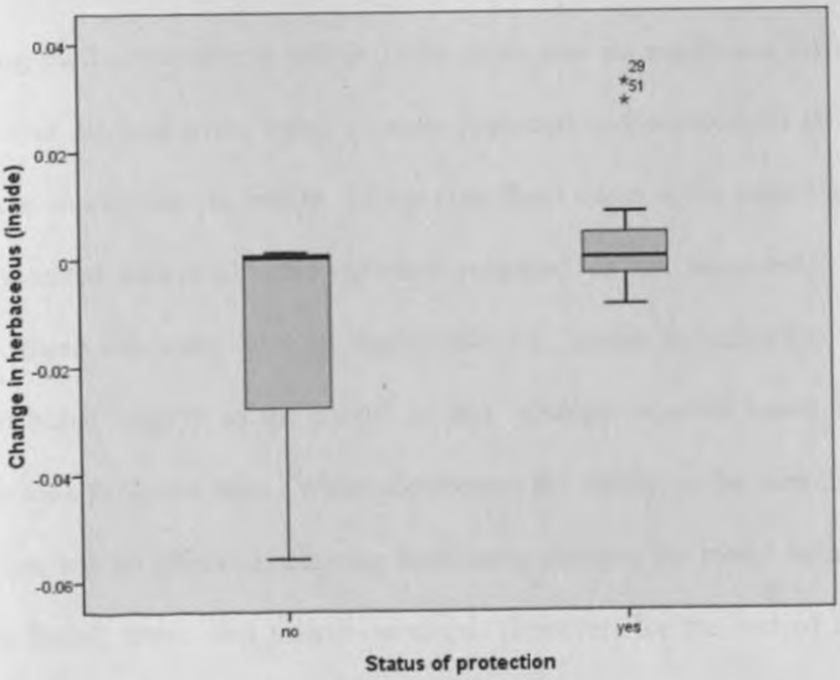

Figure 32: Box plot showing the land cover change in herbaceous INSIDE unprotected and protected IBA sites

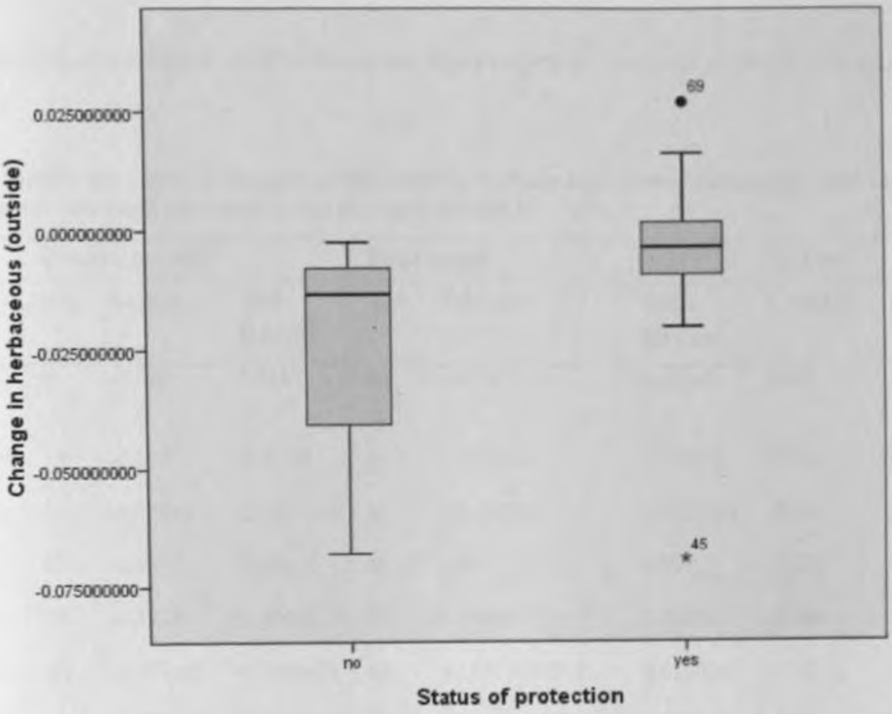

Figure 33: Box plot showing the land cover change in herbaceous OUTSIDE unprotected and protected IBA sites

When considering land cover change within IBAs, there was no significant difference in land cover change across all land cover types between protected and unprotected IBA sites (Table 24). This therefore means that protection did not contribute much to the model and hence land cover change occurred across all sites whether protected or not protected. These results notwithstanding, there was some level of significance for urban, an indication that this land cover type contributed slightly to the model in that change in urban cover was greater at unprotected sites than protected sites. When considering the change in the surrounding area and whether protection has an effect in reducing land cover change, the model returns significant results for open forest, arable and plantation crops. However, for the rest of the land cover change, the relationship is not significant (Table 26).

a) Land cover change inside and outside the IBAs unprotected and protected areas

Table 24: T-Test results for rates of change (1986-2008) in various land cover categories between protected and unprotected areas (without considering the changes around it)

	Unprotected			Protected			T-Test	
Land cover	**n=**	**Mean**	**Std. Error**	**n=**	**Mean**	**Std. Error**	**t value**	**p value**
Δ closed forest (in)	4	-0.024	0.0144	30	-.0702	0.03641	0.457	0.651
Δ closed forest (out)	15	-0.155	0.08901	34	-.2390	0.05892	0.788	0.435
Δ open forest (in)	11	-0.23916	0.1231205	39	-.015471	0.0137058	-3.254	**0.002**
Δ open forest (out)	22	-0.17	0.075	39	-.08	0.037	-1.226	0.225
Δ nat-agr mosaic (in)	19	-0.1524	0.08675	35	-.1088	0.05074	-0.464	0.644
Δ nat-agr mosaic (out)	25	-1.51E-02	0.0086626	43	-2.8589051E-2	0.0120065	0.787	0.434
Δ shrub (in)	20	-1.49E-01	0.0710278	42	-3.7381431E-2	0.034136	-1.606	0.114
Δ shrub (out)	25	-3.75E-02	0.0219101	44	-4.6424668E-3	0.0160157	-1.221	0.226
Δ herbaceous (in)	19	-0.01	0.007	40	-.03	0.025	0.348	0.729
Δ herbaceous (out)	24	-1.76E-02	0.00652	43	-1.902778E-2	0.0128779	0.082	0.935
Δ tree crop(in)	7	-0.1799	0.1615	19	-.0733	0.05781	-0.79	0.437
Δ tree crop(out	11	-0.08	0.092	27	-.07	0.052	-0.156	0.877
Δ agric (in)	22	-0.03	0.046	35	-.03	0.044	-0.038	0.97
Δ agric (out)	26	-1.98E-02	0.0396	42	-1.920174E-2	0.0272191	-0.013	0.99
Δ water (in)	20	0.0006	0.0033	12	-.0325	0.04289	0.998	0.326
Δ water (out)	20	-5.30E-02	0.0499	23	-8.078093E-2	0.0607106	0.347	0.73
Δ Flooded vege (In)	17	-0.08	0.058	14	-.17	0.1	0.826	0.416
Δ Flooded vege (out)	18	-1.18E-01	0.0758766		-1.23410997348E-1		0.058	0.954
Δ Urban (in)	2	.0341	0.03413	6	-.4740	0.23563	1.18	0.283
Δ Urban (out)	9	-.0640	0.149	17	-.15	0.1	-0.384	0.704
Δ Bare (in)	10	-.2135	0.13187	13	-.2294	0.12264	0.088	0.931
Δ Bare (out)	11	-.21	0.04753	18	-.2532	0.10069	1.403	0.172

Table 25: T-Test results for rates of change (1986-2008) in various land cover categories within protected areas

Land cover	t-value	df	p value
All forests	-1.0491	10.317	0.3181
Closed forest	1.1815	31.363	0.2463
Open forest	-1.8056	10.249	0.1004
Mosaic	-0.4336	30.529	0.6676
Shrub land	-1.4159	28.096	0.1678
Herbaceous	0.4934	44.632	0.6242
Tree crop	-0.6218	7.594	0.5523
Arable	-0.0397	50.299	0.9685
Water	0.7684	11.129	0.4582
Flooded veg	0.7917	21.291	0.4373
Urban	2.1341	5.201	**0.0838**
Bare ground	0.0882	20.051	0.9306

Table 26: Linear model results for rates of change (1986-2008) in various land cover categories inside IBAs and protected area status (land cover changes around sites considered)

Land cover	Para. Estimate	SE	F	Pr(>|t|)
Closed forest	-0.04721	0.13207	0.08739 ($F_{1,25}$)	0.9166
Open forest	0.13634	0.05565	8.32 ($F_{1,44}$)	**0.0009**
Mosaic	0.03891	0.09322	0.9829 ($F_{1,51}$)	0.3812
Shrub land	0.11582	0.06987	2.224 ($F_{1,58}$)	0.1173
Herbaceous	-0.023084	0.037666	1.404 ($F_{1,55}$)	0.2542
Tree crop	0.1985	0.1342	2.889 ($F_{1,16}$)	**0.0849**
Arable	0.04333	0.06198	6.909 ($F_{1,53}$)	**0.0022**
Water	-0.044413	0.040191	0.6612 ($F_{1,24}$)	0.5254
Flooded veg	-0.09215	0.13086	1.691 ($F_{1,22}$)	0.2075
Urban	-1.0352	0.7170	1.305 ($F_{1,3}$)	0.391
Bare ground	0.14039	0.14786	0.531 ($F_{1,11}$)	0.6023

b) Land cover change outside IBAs

Land cover change in the surrounding area was also not affected by protection in that protection or lack of protection did not prevent land cover change around these IBA sites (Table 27).

Table 27: T-test results for rates of change (1986-2008) in various land cover categories outside IBAs and outside protected area status

Land cover	t-value	df	p value
Closed forest	0.7866	26.775	0.4384
Open forest	-1.0953	31.475	0.2817
Mosaic	0.9099	65.871	0.3662
Shrub land	-1.211	48.734	0.2317
Herbaceous	0.102	59.172	0.919
Tree crop	-0.1482	16.731	0.884
Arable	-0.0127	47.733	0.99
Water	0.3535	40.409	0.7256
Flooded veg	0.0573	36.094	0.9546
Urban	-0.3745	15.286	0.7132
Bare ground	1.6986	23.442	0.1026

Protected areas showed a varied degree of land cover change in terms of the rates of change and the total extent of change between 1972 and 2008 while the percentage of natural land cover transformed to other land uses also varied from site to site.

4.5.5. Discussion

The result from t-test analysis suggests that land cover change was no less frequent at protected sites compared to those sites unprotected. Results from this study therefore suggest that land cover change is taking place across all major land cover types at both protected area sites and non-protected areas. However, compared to other land cover types, protection seems to be effective in preventing urbanization (including human settlement) in protected areas than in non-protected areas. Protection is also important for open forests and shrub lands. This is contrary to previous studies that indicate that protection has been very effective in reducing deforestation and biodiversity loss (Sánchez-Azofeifa, 1999; Nepstad *et al.*, 2006; Nagendra, 2008; Brooks *et al.*, 2009)

However, when land cover change in the surrounding area is included in the model, land cover change for open forests, arable and plantation crops is more pronounced at unprotected areas than at protected areas. However, there is no significant difference in land cover change in closed forest, natural agricultural mosaic, shrub land, herbaceous, water, urban and flooded vegetation. Protected areas are the backbone of international site based conservation initiatives. For protected areas to remain functionally relevant they should be able to achieve the objectives for which they were established (i.e. protecting species and ecosystems). Therefore the level of habitat loss at those sites should ideally be lower compared to non-protected areas but this topic has elicited much debate (Chomitz, 2007; Nelson and Chomitz, 2009). Even though the exponential increase in the number and coverage of protected areas worldwide represents conservation success, many protected areas are ineffectively managed (Ervin, 2003a). Failure to prevent land cover change, the structure and functions of these ecosystems become compromised (Vitousek *et al.,* 1997, Sala *et al.,* 2000). This means that not all protected area sites are effective in addressing the management objectives of protecting species and ecosystems and therefore the degree of effectiveness varies from country to country and site to site. For example, deforestation within officially designated protected areas varies widely, from negligible in Costa Rica in 1986–97 (Sánchez-Azofeifa *et al.*, 1999) to a decline of >56 per cent in the protected lowland forests of Indonesian Borneo in 1985–2001 (Curran *et al.*, 2004).

Establishing protected areas and applying stricter laws and greater enforcement is central in managing biodiversity at protected areas. However, in reality, this may not be the case since in most cases, protected areas may not be able to be effective in preventing biodiversity loss, cushion against land degradation and ameliorate climate change (at local, regional and global level) due to several challenges, key among them; insufficient financing, insufficient strategic planning to ensure adequate ecological and representation of biodiversity components, limited capacity of many protected areas to ensure long-term persistence of these conservation areas in terms of resources and capacity. In a study on a protected area, Liu *et al*., (2001) found out that rates of habitat loss inside the reserve increased similarly or more than outside the protected area just a few years after it was created. This contradicts a study by Bruner *et al.,* (2001), which provided evidence that protected area designation is an effective means of stopping habitat loss.

Consequently the GTS species supported within the IBAs studied are not immune to direct human land cover change. Conservation of these sites is essential for the long term persistence of these species. The results from previous studies are equivocal, with some suggesting that protected areas reduce land cover change, while others find less evidence for this conclusion. The current analysis is among the first to consider changes in land cover other than just forest. For example, in a study conducted in Mexico, out of the 69 protected areas studied, only 54% of them were effective in preventing land cover change (Figueroa and Sanchez-Cordero, 2008). The study was based on a simple comparison of protected and non protected sites. Perhaps a different conclusion would have been drawn if the sites were better matched by controlling the location of protected areas and their characteristics (e.g. comparing protected areas points with similar unprotected points, controlling for slope, rainfall, road proximity (Chomitz, 2007; Nelson and Chomitz, 2009; Joppa and Pfaff, 2009b, 2009c; Pfaff *et al.,* 2009). The non-significant relationships and results from correlating rates and extent of land cover change with protection could also be attributed to the design of the study and the assumptions that all protected areas regardless of the category enjoy the same level of protection and management effectiveness. For example, in this study, aggregating all protected area categories into one group could be erroneous because some of the protected areas (e.g. National Parks) are strictly no go zones and meant for conservation while Game Reserves permit a certain level of human activities. This could be fuelling the observed levels of land cover change at some of the protected area sites. Moreover, with the paradigm shift from the historical exclusionist approach to a community-based natural resource management and permitting some level of extractive uses of natural resources at these protected areas , the latter approach if not well controlled may become one way through which land cover change will surreptitiously continue to be lost at key biodiversity sites.

Understanding land cover and land use dynamics outside protected areas or key biodiversity sites is critical. A study by DeFries *et al.,*(2010) recommend that protected areas embedded in human-dominated landscapes require improved understanding of the ecological processes affected by land use in the surrounding landscapes and the first step should be to delineate the zone of interaction around parks that encompass the hydrologic, ecologic, and socioeconomic interaction between parks and their surroundings. For example, various studies on land cover change inside and outside demonstrate the effectiveness of protected areas in reducing land cover change (Chatelain *et al.,* 1996; Sánchez-Azofeifa, 1999; Beresford *et al.,* 2012 *In prep*). A study of fragmentation and deforestation inside and outside of protected areas in the Sarapiqui region of Costa Rica showed that protected areas were dramatically curtailing the rate of deforestation.

Most shrub land and grassland protected area IBAs confined to savannah and grassland ecosystems are embedded in a sparsely populated matrix save for those found near urban areas (e.g. Nairobi National Park IBA, Kenya). For protected areas that are found in these marginal areas, the protection status accords such a site a lot of security thus easing pressure. Most such sites apart from those in human dominated landscapes are therefore not facing a lot of anthropogenic pressures probably because some of them are found in remote places, rugged terrains and in areas hostile to crop husbandry as a result of poor soils and low rainfall.

However, this is not the case for sites that are not protected and which are in human dominated agricultural landscapes. These pressures and continued land cover change manifested through transformation of land from natural vegetation to arable unless controlled will have a significant impact on ecosystems and the species dependent on these sites. The fact that some of the species at some of these sites are range restricted, endemic and globally threatened (e.g. the Hinde's babbler in the Mukurueni Valley IBA, Machakos Valley IBA and Kianyaga Valley IBA or Sharpe's Longclaw at the Kinangop highland grasslands) makes them of global conservation concern, thus invariably consigning such species to the vagaries of extinction.

In this study, and considering the effect of leakage, protection did not lead to significant deleterious land cover change in the surrounding matrix. This could be attributed to the fact that protection may not be water tight to an extent that there is absolute exclusion of human activities or encroachment. The porous boundaries in some protected areas as well as the low level of management effectiveness and enforcement at some sites does not therefore entirely transfer or displace conversion and modification to the unprotected surrounding matrix. Even though there has been a holistic effort to focus on conserving wildlife beyond boundaries, this has not been pursued dexterously. Additionally, protected area managers have only concentrated their conservation efforts within the areas of their jurisdiction. There is an urgent need for them to be also concerned with the land cover and land use dynamics outside the protection area boundaries because of the potential impact of these dynamics on the integrity of the protected area itself.

4.6. Are Site-based conservation approaches effective? (ii): Land cover change and site specific past conservation interventions

4.6.1. Introduction

With increasing threats at key biodiversity sites, various actors are intervening to respond to the pressure on natural resources and reverse the trends in land cover change as well as biodiversity loss. Interventions come in various forms and approaches, various hierarchical levels and actors including land and water protection (e.g. terrestrial and marine protected areas), species management, education and awareness, law and policy, livelihoods, economic and other incentives and capacity building for conservation (Brooks *et al.,* 2009). Site-based conservation interventions are widely recognized as being important in tackling global biodiversity loss (Joppa, *et al.*,2009a; Nagendra, 2008) being implemented at various priority sites to promote biodiversity conservation and maintain ecological integrity of ecosystems, which provide ecosystem services, livelihoods and sustenance to local communities (IUCN, 2005). Some of the interventions include designation of protected areas (Dudley, 2008), conservation and management planning, environmental education, political lobbying, restoration and rehabilitation of degraded ecosystems, reforestation, monitoring, demarcation of boundaries, eradication of alien invasive species, and prevention of wild fires and climate change mitigation and adaptation measures.

Governmental organisations, non-governmental organisations, civil society organisations, private sector actors, community-based groups, development partners are investing substantially in biodiversity conservation and livelihood interventions at selected priority sites, habitats landscapes and ecosystems. Ideally, effective conservation interventions would prevent loss of or maintain the stability of biodiversity and therefore such interventions are deemed effective if they slow down the human-induced rate of biodiversity loss (Rodrigues, 2006) at a species and land cover level. Even though biodiversity conservation is the primary goal (Hughes and Flintan, 2001), this participatory and collaborative approach integrates both economic and empowerment benefits to local communities, permits practices that embrace different values and local knowledge of the biological resources and human activity (Fischer, 1995; Borrini-Feyerabend, 1996). According to Hughes and Flintan (2001), this collaborative approach is based on two assumptions. First, diversified local livelihood options will reduce human pressure on biodiversity leading to its improved conservation. Secondly, local people and their livelihood practices, rather than 'external factors', comprise the most important threat to the biodiversity resources of the area in question.

As a paradigm shift, Community-based Natural Resource Management concept has evolved over the years as a mechanism for rural development, local empowerment and biodiversity conservation (Roe *et al.,* 2009). This approach takes many different forms in different locations and different socio-political and biophysical contexts (Mulder and Coppolillo, 2005). For example, for many years, BirdLife International has worked with community-based groups (hereby referred to as Site Support Groups or Local Conservation Groups) at some of its IBAs as a mechanism to conserve natural resources while improving people's livelihoods (BirdLife International, 2010b). These groups participate in an array of conservation and livelihood improvement initiatives while also working with other stakeholders including government agencies and the private sector. Tangible activities that promote conservation are very diverse and include restoration of degraded habitats, research and monitoring, enforcement of laws and policy, environmental education and awareness raising and eradication of alien invasive species (BirdLife International, 2007a). The measure of the success and the impact of SSGs is the best way to evaluate the effectiveness of conservation outcomes at IBAs.

Engaging communities is a paradigm that aims at improving conservation and therefore there is need for assessment and analysis of the impact of this engagement (e.g. the extent to which the participation is translated to improved biodiversity conservation and reduced land cover change). In most cases, the impact of conservation investments on both biodiversity and livelihoods is not only not well assessed but is also not simple to evaluate leading to uncertainties about the contribution of conservation actions in reducing extinction rates (Brooks *et al.,* 2009). Evaluating the positive impact of these approaches on biodiversity conservation is needed to ascertain whether the goals of saving species from extinction and protecting sites and habitats from detrimental activities are being achieved. While the effectiveness of Protected Areas are becoming better known, few studies have attempted to evaluate the effectiveness of site-based conservation approaches based on working with communities in integrated conservation and livelihood improvement initiatives. In order to maintain, replicate, improve, adapt, and adopt new mechanisms and approaches in the efforts of working with communities through programmes and projects, there is need to track and assess the effectiveness of these initiatives in preventing land cover change at sites. IBAs are a useful suite of sites to examine, given that some have community based conservation measures and others do not. Investigating the relationships between site-based interventions and land cover change is critical in justifying the continued financial investments made in time and space while at the same time providing an impetus for replication of these interventions at other priority sites in the region or at a global level. The efficacy of these approaches in stemming current rates of biodiversity loss will stimulate the process for scaling up and replicating of the approaches at other sites. This study tests the effectiveness or deficiency of site-based management and conservation interventions that involve working with communities at reducing land cover change at these sites.

4.6.2. Aims

The aim of this section is to assess the effectiveness of past site-based conservation interventions by BirdLife International at respective IBAs in preventing deleterious land cover change and biodiversity loss. Particularly, focus was on how effective the BirdLife International model of working with a network of SSGs and implementing actual site conservation projects has been in averting biodiversity loss.

The following hypotheses are tested in this section:

a) *Land cover change is less frequent following long-term site-based conservation initiatives.*
b) *The rate of and total extent of change is lower for sites with long-term site-based conservation interventions*

4.6.3. Methods and Analysis

Of the 72 IBA sites selected for this study, information on those that had a history of long-term conservation interventions was identified, particularly where SSGs have been established and are operational as well as where these groups are non-existent. The rates of land cover change were established using the Land Cover Change Graphical User Interface (see procedure in chapter 3). Rates of land cover change for respective land cover types at selected IBAs (see selection criteria in Chapter 3) were regressed with the presence or absence of SSGs as part of past site specific conservation interventions. T-Tests were used to determine whether there has been a significant difference in deleterious land cover change at sites with SSGs or not or whether it has been less deleterious considering land cover change as a dependent variable and BirdLife SSGs as independent variables hence the presence or absence of SSGs was considered during these analyses.

4.6.4. Results

a) Rates of land cover change inside IBAs (1986-2008)

For the 72 IBA sites used in this study, Site Support Groups established between 1998 and 2006 were present at 13 sites (18.1% of the sample sites) while 59 sites (81.9%) did not have these groups. Based on the t-test results (Table 26) rates of land cover change for the various land cover types at IBAs with and without SSGs, there was no significant relationship for all apart from tree crops and arable. Apart from arable and tree crop land cover types, this study therefore provides insufficient evidence that land cover change was less frequent at sites where SSGs were present compared to those sites where SSGs were absent. This means that based on the data used, SSGs are yet to make an impact on biodiversity (i.e. reduced land cover change). Considering the rates of change in all forest habitats, there was no significant difference in forest loss on sites with SSGs compared to sites with no SSGs, hence an indication that based on this study, SSGs as an intervention did not have any influence on preventing detrimental change in land cover. The same pattern was observed for other land cover types including closed forest, open forest, natural-agricultural mosaic, shrub land, grasslands and wetlands respectively.

Table 28: T-Test results showing the differences for rates of change (1986-2008) in various land cover categories sites with and those without SSGs

	no SSGs			SSGs			T-Test	
Land cover	**n=**	**Mean**	**Std. Error**	**n=**	**Mean**	**Std. Error**	**t value**	**p value**
Δ closed forest (in)	26	-.0756	0.04184	8	-.0295	0.01472	-0.602	0.552
Δ closed forest (out)	42	-.2301	0.05583	7	-.1121	0.06561	-0.841	0.405
Δ open forest (in)	38	-.061332	0.031903	12	-.07529	0.084493	0.19	0.85
Δ open forest (out)	52	-.13	0.042	9	.00	0.015	-1.222	0.226
Δ nat-agr mosaic (in)	42	-.1402	0.05196	12	-.0679	0.08538	-0.672	0.505
Δ nat-agr mosaic (out)								
Δ shrub (in)	51	-6.9667097E-2	0.035264	11	-9.05719E-2	0.091326	0.241	0.81
Δ shrub (out)	57	-2.4860404E-2	0.014428	12	-.02	0.027996	-1.405	0.165
Δ herbaceous (in)	48	-.02	0.021	11	2.29209-2	0.013	-0.185	0.854
Δ herbaceous (out)	56	-1.9435643E-2	0.010173	11	-1.37382E-2	0.005969	-0.245	0.807
Δ tree crop(in)	19	-.1531	0.07778	7	.0368	0.0322	-1.448	0.161
Δ tree crop(out	28	-.10	0.06	10	.02	0.008	-1.214	0.233
Δ agric (in)	45	-.06	0.038	12	.07	0.042	-1.6	0.115
Δ agric (out)	55	-2.715652E-2	0.027668	13	.07	0.00442	-0.705	0.483
Δ water (in)	29	-.0130	0.01771	3	.0000	0.00148	-0.232	0.818
Δ water (out)	37	-7.84883E-2	0.04576	6	-2.33449	0.001146	-0.663	0.511
Δ Flooded vege (In)	27	-.10	0.053	4	-.26	0.248	0.947	0.352
Δ Flooded vege (out)	36	-1.3895107E-1	0.054897	5	9.10302E-3	0.019667	-0.993	0.327
Δ Urban (in)		-.3469			.000			
Δ Urban (out)		-.22			.01			
Δ Bare (in)		-.1960			-.5000			
Δ Bare (out)		-.1651			-.3231			

Table 29: T-Test results for differences in rates of change (1986-2008) in various land cover categories when SSGs are present or absent (without considering the changes around it)

Land cover	t-value	df	p value
All forests	-1.0768	21.916	0.2933
Closed forest	-1.0398	29.933	0.3068
Open forest	0.1545	14.274	0.8793
Mosaic	-0.7227	19.924	0.4782
Shrub land	0.2135	13.146	0.8342
Herbaceous	-0.3347	53.484	0.7391
Tree crop	-2.2559	22.698	**0.0340**
Arable	-2.1945	31.857	**0.0356**
Water	-0.729	28.375	0.472
Flooded veg	0.6135	3.282	0.5795
Bare ground	0.5988	1.063	0.6517

When looking at absolute changes, SSG intervention was only effective for change in tree crops and arable land cover. This could mean that SSGs are effective in preventing cash crop/plantation crop increase and arable increase at IBAs. This is a positive effect on biodiversity at these sites.

4.6.5. Discussion

A paradigm shift towards community-based natural resource management as an alternative effective approach to preventing ecosystem degradation and species extinctions has gained considerable momentum and popularity (Bray *et al.*, 2003; Kothari, 2006). This is partly because the poor are dependent on biodiversity for their day-to-day livelihoods while biodiversity conservation can be a means for poverty reduction (Thompson *et al.*, 2007, Secretariat of the Convention of Biological Diversity, 2008). Since most drivers of biodiversity loss are attributed to anthropogenic causes, embracing community-based natural resource management is an approach, which is gaining popularity and an increasingly important facet of conservation policy and practice linking biodiversity conservation and poverty alleviation (IUCN, 2005). This is even more imperative and urgent given protected areas dependency on the surrounding matrix to maintain flow of biota, water, nutrients and energy yet protected area managers have little authority over the surrounding landscape where land use and interactions can have major impacts on the integrity of the protected area (DeFries *et al.,* 2010) or key biodiversity site.

Linkages between biodiversity conservation and poverty reduction have been demonstrated to some degree (O'riordan and Stoll-Kleemann, 2002; Secretariat of the Convention on Biological Diversity, 2008; BirdLife International, 2010a). However, demonstrating the impact of these poverty alleviation interventions on biodiversity conservation and vice versa is a challenge (Kepe *et al.,* 2004, Thompson, *et al.,* 2007). According to Rodrigues (2006), measuring conservation impact is not simple as biodiversity is not easily quantified.

This study fails to demonstrate a consistent pattern of improved conservation outcomes emanating from engaging SSGs. There is no empirical evidence that the SSG approach has made any tangible impact in the form of better conservation outcomes as far as conserving biodiversity is concerned across all major land cover types studied apart from shrub land IBAs. Neither is there evidence that SSGs are contributing to stemming deleterious land cover change outside IBAs. Other studies however support these findings. For example, studies expressed the difficulties proving that participatory conservation generates sustained success (Wells and

Brandon, 1992). In contrast, other studies have demonstrated a reduced extent of illegal resource use and steady increase in species populations (e.g. Plumptre, 1998). Monitoring and evaluation efforts have demonstrated that the SSG concept is working and is effective in reducing deleterious land cover change (e.g. through restoration and reforestation initiatives, Thompson *et al.*, 2007; BirdLife International, 2010a, b).

However, this study and the results depicted may be attributed to the design and methodological issues employed in this study as well as the fact that SSG approach is a new concept and it will require a longer time before the impact is felt at site level. For example, the SSG approach is a new concept being applied at IBAs and in most cases, the period for which these groups have been in operation at these sites varied considerably with most ranging from to two to ten years. Some of the interventions take longer to actually demonstrate tangible conservation impact whereas, the impact on livelihood projects on biodiversity still remains unclear. This means that most of them have invariably been in operation after 2000 yet the land cover change analysis focuses on the years 1986 to 2008; hence it is too soon to expect their interventions to have slowed down the rates of land cover change in most land cover types at IBAs.

It is therefore important that the impact of SSGs work on reducing land cover change be investigated further in future while using the recent years during which SSGs have been at the sites as periods of focus. In addition, even though this correlation does not return significant relationships, this may not be an indicator that SSGs are not effective. Some of the SSGs are involved in diverse activities some of which are focussed on single species conservation and therefore comparing the status and trends in population of these species would be useful in demonstrating the impact of their efforts. For example, there are species, which have adapted to new modified habitats and therefore populations may be stable or at least not declining (e.g. Hinde's Babbler with the *Lantana camara* shrub land in a human dominated landscape IBA) and therefore even though there may have been a decline in the bush land, the remaining Lantana thickets may be a refuge for them. This could be one reason why the impact of SSGs in reducing land cover change for shrub land IBAs appears significant. In view of this assessing land cover change, community conservation and trends in populations of trigger avian species would add value to assessing conservation outcomes attributed to working with communities (e.g. SSGs).

However, the insignificant relationships accruing from this study could be attributed to the small sample size of SSGs used in the analyses. For example, out of the 72 sites, using only 13 sites for the analysis may have introduced errors because of the small sample size of IBAs. Moreover, most of the IBAs were found at forest IBAs more than in other land cover types. Out of the 13sites with SSGs, eight were forest sites, two wetlands, two were natural-agricultural mosaic and one was artificial landscapes. This therefore means that for each land cover type, the sample size of the SSGs was not sufficient and representative in making a statistically robust conclusion and as such a larger sample of SSGs would be needed in future analyses. In terms of the results from the correlations between SSGs and land cover change outside IBAs, the relationship is also not significant. This could be as a result of the sample size as well as the fact that the main remit of SSGs is on the conservation and protection of IBAs and not the conservation of biodiversity outside IBAs. However, it is anticipated that with the emphasis on the importance of biodiversity in the matrix surrounding IBAs, the focus of SSGs will diversify their intervention activities in a way that their activities will also contribute to the conservation of natural vegetation outside

IBAs. This is very critical especially in a human dominated and fragmented landscape where species have to move across the matrix to other natural habitats. This could be achieved through sustainable environmental friendly land use systems and best practices such as embracing agroforestry and other sustainable nature-based enterprises that will boost agricultural biodiversity and ecological processes surrounding landscapes. This if done will promote linkages in the landscape and provide corridors and connectivity for species mobility within a fragmented surrounding landscape. Above all, it will increase agricultural productivity, reduce dependency and ease the pressure and deleterious land cover change on the key biodiversity sites.

However, there is need to develop and apply methods and tools for tracking the impact of working with community groups at IBAs on biodiversity and livelihoods as well as develop baselines against which future assessments will be based. For example, Roe, *et al.*, (2009), report that a major deficiency of formal community-based natural resource management systems and programmes is the absence or paucity of quantitative and/or qualitative data on their social, economic and environmental impacts. Monitoring in the long term, the species and ecological processes being targeted for conservation, there won't be any reliable evidence to show the impact of SSGs on biodiversity conservation. Lack of documented examples of success in community based conservation has left people to question whether this approach actually works and whether it is easy to demonstrate causative links between conservation and development (Hughes and Flintan, 2001). Documentation to prove these links is very critical in guiding BirdLife International on how to review the SSG strategies and approaches to maximize on conservation outcomes and livelihood improvements. In general there is need to continue working with communities, distilling and sharing lessons learnt from past engagement with communities in projects and programmes (Clay, 1996; Larson *et al.*, 1998; Thompson *et al.*, 2007).

4.7. Can one predict which IBAs are at greatest risk of deleterious land cover change?

4.7.1. Introduction

Drivers of land cover change tend to act singly or synergistically, directly or indirectly to exacerbate deleterious impacts on biodiversity at various hierarchical scales as a result of cascading and cumulative effect of these threats (Brook *et al.*, 2008). In view of these interactions, it is imperative that sites that are known to be at high risk of deleterious land cover change be identified and mapped as hotspots to facilitate timely and appropriate solutions and mitigative measures to avert and control drivers and ameliorate the accelerated rate of land cover change. This is even more important if the right interventions, strategic conservation decisions and investments are to be made timely and cost-effectively to achieve greater conservation outcomes at site or landscape level.

Lack of understanding of the complex human–environment interactions and the synergistic effects of threats to biodiversity is one of the challenges to effective conservation (Underwood *et al.*, 2009). At broad levels, hotspots have been identified across the world as the richest and most threatened reservoirs of plant and animal life on earth (Mittermeier *et al.*, 2004). However, with limited resources, there is also need to identify these areas at a local level so that systematic conservation planning and investment priorities can be initiated. Even for IBAs, while identifying these sites is a priority and has been one of the most successful programmes by

BirdLife International, very few systematic studies and analyses have been undertaken for IBAs incorporating threats. Where this has been done, it is largely based on the presence of globally threatened birds and not on interactions between the increasing types of threats.

4.7.2. Aims

Based on the analysis of land-cover dynamics over the last two decades for IBAs, the study aims to predict IBAs that are at greatest risk of detrimental land cover change. This is achieved by identifying driving forces for land cover change that act in combination to exert adverse pressures and how these threats make sites and species more vulnerable.

4.7.3. Methods and Analysis

Based on the study questions, pressures at IBAs that were considered under this study are two, namely human population density and agricultural intensity. In this section, a multiple regression done in R software was used to determine the relationship between several independent or predictor variables and a dependent variable. In this case, the independent variables were population density and agricultural intensity around the IBAs while land cover change was the dependent variable. The level of significance was interpreted to show the strength or contribution of each factor or variable on the model. All of the above results aggregated into an overview analysis and produce a statistical model that identifies which (if any) characteristics make IBAs more vulnerable. The various land cover types were considered in this model to assess and show which are most vulnerable as a result of the combined nature of the drivers of land cover change.

4.7.4. Results

a) Rates of land cover change inside IBAs (1986-2008) and combined threats

The relationship between rates and extent of cover change during 1986-2008 correlated with a combination of human population density and agricultural intensity around IBAs was assessed. The relationship was significant for all forest and herbaceous IBAs but not significant for the rest of the land cover types. These relationships are presented in Tables 30-37. The results show that that without any conservation intervention (protection and SSGs), the effect of these threats are very significant for all natural forests, open forests, shrub land and grassland (herbaceous) land cover types (Tables 30, 32, 34 and 35). This therefore suggests that there is a significant change in land cover for these habitats when these threats interact. However of the two threats, agricultural intensity has a strong effect on the model when assessing the relationship for all natural forest habitats (Table 30), open forest (Table 32), shrub land (Table 34) and grassland (Table 35). This suggests that when these two threats (agricultural intensity and population density) are interacting, population density doe is not significant across all land cover types.

Table 30: Combined effect of agricultural intensity and population density as threats on all natural forest land cover change

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-2.494e-02	1.575e-02	-1.583	0.12169
Agric intensity	-6.042e-03	2.196e-03	-2.752	**0.00903 ***
Pop. density	-7.086e-06	1.603e-05	-0.442	0.66094

R^2= 0.1548, $F_{(2, 38)}$ = 4.662, p= **0.01546**

Table 31: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on closed forest land cover change at IBAs

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-8.212e-02	5.201e-02	-1.579	0.127
Agric. intensity	5.672e-03	8.434e-03	0.673	0.507
Pop. density	-1.100e-05	4.789e-05	-0.230	0.820

R^2= -0.06019, $F_{(2, 25)}$ = 0.2336, p= 0.7934

Table 32: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on open forest land cover change at IBAs

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-9.895e-03	4.765e-02	-0.208	0.8366
Agric. intensity	-1.256e-02	6.728e-03	-1.866	**0.0695**
Pop. density	-3.013e-05	4.944e-05	-0.609	0.5458

R^2= 0.06797, $F_{(2, 39)}$ = 2.495, p= **0.09558**

Table 33: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on natural-agricultural mosaic land cover change at IBAs

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-1.372e-01	7.140e-02	-1.922	0.0608 .
Agric. intensity	-4.157e-03	8.992e-03	-0.462	0.6460
Pop. density	7.467e-05	1.348e-04	0.554	0.5825

R^2= -0.03481, $F_{(2, 46)}$ = 0.1926, p= 0.8255

Table 34: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on shrub land cover change at IBAs

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-2.499e-02	5.398e-02	-0.463	0.6456
Agric. intensity	-1.533e-02	7.199e-03	-2.129	**0.0386** *
Pop. density	7.595e-05	1.066e-04	0.712	0.4798

R^2= 0.05016, $F_{(2, 46)}$ = 2.267, p= 0.1150

Table 35: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on herbaceous land cover change at IBAs

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	2.032e-02	2.703e-02	0.752	0.45630
Agric. intensity	-1.297e-02	4.571e-03	-2.837	**0.0069** **
Pop. intensity	3.317e-05	3.083e-05	1.076	0.28796

R^2= 0.1185, $F_{(2, 43)}$ = 4.025, p= **0.02498**

Table 36: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on water land cover change at IBAs

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-1.673e-02	2.424e-02	-0.690	0.497
Agric. intensity	6.556e-04	4.469e-03	0.147	0.885
Pop. density	-2.714e-06	2.610e-05	-0.104	0.918

R^2= -0.08217, $F_{(2, 24)}$ = 0.01291, p= 0.9872

Table 37: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on flooded vegetation land cover change at IBAs

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-1.340e-01	1.129e-01	-1.188	0.248
Agric. intensity	5.120e-03	2.499e-02	0.205	0.840
Pop. density	-8.063e-05	3.906e-04	-0.206	0.838

R^2= -0.08766 $F_{(2, 22)}$ = 0.03286, p= 0.9677

However, when sites are subjected to agricultural intensity and population density and then interventions (protection and SSGs) introduced into the model, land cover change is significant for all natural forests (Table 38), open forest (Table 40) and grassland/herbaceous (Table 42). However, agricultural intensity is still a significant factor in the model as far as land cover change in all natural forests and grassland/herbaceous is concerned. Protection is not a significant factor in all land cover types apart from open forest land cover (Table 40). This suggests that protection is important in reducing land cover change for open forests (e.g.

woodlands). Interventions in the form of SSGs still have no effect when the two threats are interacting at sites that are protected and where the SG approach is being applied.

Table 38: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on all natural forests land cover change

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-6.048e-02	2.887e-02	-2.095	0.0433 *
Agric. intensity	-5.719e-03	2.189e-03	-2.612	**0.0130** *
Pop. density	-1.087e-05	1.590e-05	-0.684	0.4985
Protection	2.950e-02	2.725e-02	1.082	0.2863
SSGs	4.247e-02	2.560e-02	1.659	0.1058

R^2= 0.1877, $F_{(4, 36)}$ = 3.311, p= **0.02076**

Table 39: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on closed forest land cover change

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-6.589e-02	1.245e-01	-0.529	0.602
Agric intensity	4.392e-03	9.445e-03	0.465	0.646
Pop. density	-1.513e-05	5.062e-05	-0.299	0.768
Protection	-2.442e-02	1.153e-01	-0.212	0.834
SSGs	3.807e-02	8.724e-02	0.436	0.667

R^2 =-0.1415, $F_{(4, 23)}$ = 0.1631, p= 0.955

Table 40: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on open forest land cover change

	Para.Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-1.801e-01	8.001e-02	-2.251	0.0304 *
Agric intensity	-1.032e-02	6.422e-03	-1.607	0.1165
Pop density	-3.305e-05	4.722e-05	-0.700	0.4883
Protection	1.988e-01	7.731e-02	2.572	**0.0143** *
SSGs	4.547e-02	7.549e-02	0.602	0.5506

R^2 =0.1693, $F_{(4, 37)}$ = 3.088, p-value: **0.02728**

Table 41: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on natural agricultural-mosaic land cover change

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-2.026e-01	1.156e-01	-1.753	0.0865
agric intensity	-2.381e-03	9.525e-03	-0.250	0.8038
Pop. density	5.147e-05	1.390e-04	0.370	0.7129
Protection	6.132e-02	1.116e-01	0.549	0.5854
SSGs	8.671e-02	1.179e-01	0.735	0.4660

R^2=-0.06187, $F_{(4, 44)}$ =-0.3008, p= 0.8759

Table 42: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on shrub land cover change

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-6.677e-02	8.407e-02	-0.794	0.4313
Agric. intensity	-1.422e-02	7.478e-03	-1.901	**0.0638**
Pop. density	7.006e-05	1.108e-04	0.632	0.5305
Protection	6.239e-02	8.696e-02	0.717	0.4769
SSGs	-1.811e-02	9.732e-02	-0.186	0.8532

R^2= 0.01894, $F_{(4, 44)}$ = 1.232, p= 0.3113

Table 43: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on herbaceous land cover change

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	5.952e-02	4.572e-02	1.302	0.20029
Agric intensity	-1.453e-02	4.728e-03	-3.074	**0.00375** **
Pop. density	3.256e-05	3.126e-05	1.042	0.30358
Protection	-5.725e-02	4.629e-02	-1.237	0.22318
SSGs	2.998e-02	4.965e-02	0.604	0.54921

R^2= 0.1145, $F_{(4, 41)}$ = 2.454, p=**0.06088**

Table 44: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on water land cover change at IBAs

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	3.155e-03	3.366e-02	0.094	0.926
Agric intensity	-5.056e-04	4.766e-03	-0.106	0.916
Pop. density	-9.526e-06	2.762e-05	-0.345	0.733
Protection	-4.349e-02	4.643e-02	-0.937	0.359

SSGs	3.852e-03	6.768e-02	0.057	0.955

R^2= -0.1322, $F_{(4, 22)}$ = 0.2409, p= 0.9121

Table 45: Combined effect of agricultural intensity and population density as threats and protection and SSGs as interventions on flooded vegetation land cover change at IBAs

	Para. Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-7.110e-02	1.491e-01	-0.477	0.639
Agric. intensity	1.591e-03	2.631e-02	0.060	0.952
Pop. density	1.830e-06	4.652e-04	0.004	0.997
Protection	-9.694e-02	1.485e-01	-0.653	0.521
SSGs	-1.394e-01	2.258e-01	-0.617	0.544

R^2= -0.1461, $F_{(4, 20)}$ = 0.2349, p= 0.9154

4.7.5. Discussion

All IBAs are critical for the survival of globally threatened species, range restricted, biome-restricted and congregatory species respectively (Fishpool and Evans, 2001). Based on this study, it is apparent that with agricultural intensity and population density interacting collectively as threats at IBAs, shrub land and grassland IBAs will be affected most due to deleterious land cover change. Habitat loss and human population density have been shown to predict spatial variation in the current threat status of species (Kerr and Currie, 1995; Thompson and Jones, 1999; McKinney, 2001; Brooks *et. al.,* 2002). However, as contributing to habitat loss, agricultural intensity still remains the most important factor to watch in order to avert further decline in land cover change at IBAs. This is evidenced by the impact it has on all natural forests (all forest habitats), open forests, shrubland and herbaceous land cover types. This therefore means that for these land cover types, preventing or controlling this factor is critical in securing the conservation of sites that are dominated by these land covers.

Comparatively, even though population density is not as significant a factor as agricultural intensity, it is worth noting that both factors when interacting contribute more to driving deleterious land cover change. This correlation concurs with Scharlemann *et al.*,(2005) who deduced that agricultural land use was a better predictor of threat status than human population and that average levels of threat attributable to agriculture were better predicted by land use than human population density. In most cases, agricultural intensity is driven in part by a rise in the population, which exerts pressure on the available land and encroaches onto adjacent land, transforming pristine land into agriculture. Even though, while interacting with agricultural intensity alone, population density may not contribute a lot to the observed land cover change at IBAs. However, there is need to investigate the indirect effects associated with high population density such as urbanisation, introduction and spread of alien invasive species, habitat fragmentation and increased frequencies of anthropogenic fires, which have been noted to be pervasive threats to biodiversity in other regions (Underwood *et al.*, 2009). There is therefore need for more investigation in future on the role of population density and to track its impact on biodiversity conservation at various spatial and temporal scales. Moreover, there may be other factors both direct and underlying that could be very decisive in this model, yet they were not considered as part of the analysis.

For sites that are protected and where the SSG approach is being applied, when agricultural intensity and population density interact, the cumulative effect of these threats on land cover change is still observed for all natural forests (ala forest types), open forests (most likely woodlands) and herbaceous. This suggests that even in the face of interventions, these land cover types are still facing deleterious land cover change. There seems to be less land cover change for shrub land sites when these threats are interacting where the two interventions (protection and SSGs) are being applied.

Forest, shrub land and grassland habitats or land cover still experience a lot of pressure from extensive clearance for crop production and livestock pasture, agricultural intensity (Ramankutty and Foley, 1999) and unless checked, forest cover will continue to be lost at the expense of species richness particularly forest specialists, which would not survive in any other habitat type other than forests. The forest sites are highly vulnerable most likely because forests are associated with fertile soils, adequate precipitation and high population density that provide the most favourable condition for crop husbandry. The same hypothesis may apply for grassland IBAs especially those that are found in high potential areas such as highland grasslands. As a result of this, studies have shown a level of spatial congruence between human population density and species richness (Luck, 2007). For example, many studies have documented the dramatic decline in grassland habitats (Ndang'ang'a *et al*., 2002; Wakelin and Hill. 2007) and many bird species that depend on grasslands have declined (Jansen *et al*., 1999; Temple *et al*., 1999; Vickery and Gill, 1999; Vickery *et al*., 1999). These means that these two habitats are the ones most at risk of deleterious land cover change. The implication of this would be a projected loss of forest and grassland species including extinction of endemic species and a reduction in the distribution range of certain species of global conservation concern.

In the context of these findings, it is most likely that conservation actions targeting multiple drivers of land cover change will effectively address and reduce rates of land cover change. With the emerging known and unknown impacts of climate change, there is need to incorporate the latter in conservation efforts because of the interactions with other threats to amplify the magnitude of land cover change as well as impact on the survival on species. There is therefore need to address these drivers of land cover change through integrated approaches. Investigating other latent threats to biodiversity besides the known ones will make it possible to effectively manage and protect biodiversity and key sites.

4.8. Overall Discussions

This study provides sufficient evidence that land cover change is taking place variably at different time periods, paces and degrees of magnitude and with diverse biophysical implications on both habitats and species. This is consistent with a study by Baulies and Szejwach (1998), which deduced that land use dynamics will play a major role in driving land cover changes. In this study, these changes are attributed to both proximate (extent of agriculture around IBAs) and underlying factors (human population density) besides other factors that were not part of this analysis. Correlation between land cover change for various land cover types and causative factors (e.g. agricultural extent and human population density) returns varied levels of significance, with agricultural intensity around IBAs being a strong correlate of the observed extent and rates of land cover change, while population density was less strongly associated with changes.

In other studies, correlations between deforestation and multiple causative factors revealed no distinct pattern (Rudel and Roper, 1996; Bawa and Dayanandan, 1997; Mather *et al.*,1998). Previous studies attribute agricultural expansion as one of the factors driving deleterious land cover change (Sala *et al.*, 2000; Donald, 2004; Scharlemann *et al.*, 2004). These results are consistent with a review of 152 cases on tropical deforestation, which showed that 96% of these deforestation cases were attributed to agricultural expansion (Balmford *et al.*, 2003). Elsewhere, studies have shown that intensified agriculture leads to a significant decline in bird population (Donald *et al.*, 2001).

Consequently, considering the deleterious impact of agricultural extent around key biodiversity sites on land cover change and the resultant impact on species, this threat needs to be addressed through various approaches at various hierarchical levels. However, the future course of agricultural expansion is more scenario dependent in IBAs than in the rest of the world. This is because development policies have considerable potential to either ease or exacerbate the disproportionate impact of agriculture on areas of highest biological value. There is an urgent need to reconcile the various land uses in line with the need to conserve species within their historical range. In Europe for example, policy-driven land use change had continent-wide consequences on various bird species (Green, 1996; Wilson *et al.,* 1997) and bird communities (Campbell *et al.,* 1997; Newton, 1998).

Assessing the impact of combination of threats is critical for quantifying the status of IBAs because these changes in turn trigger adverse effects on species that reside at these sites. Even though populations of certain species have been known to decline, this may partially be attributed to land cover change which affects the structure and functioning of their habitats especially for species that are restricted to specific habitats. However, there is also need to understand through further causal research into other underlying factors and ecological parameters that determine species survival. For example, to reflect the true scenario, species-habitat relationships are accurate only to the extent that habitat relationship of particular species is backed by well documented empirical data. This type of research comes in handy when population monitoring and trend analyses do not point to one certain cause (i.e. deterministic) but takes into consideration other underlying factors (life-history traits, demographic, genetic, environmental stochasticity and natural catastrophes) that may be working in synergy with land cover change to drive species into extinction. Vitousek *et al.,* (1997) recognizes the need for a clear understanding of factors driving land use requires integration of the social, economic and cultural causes of land transformation with evaluations of its bio-physical nature and consequences. Models developed without incorporating other stochastic events may be flawed. The increasing need to monitor species and land cover/use change is indisputable. Constant concurrent species-habitat monitoring efforts to keep track of both species populations and status and condition of the sites and habitats for which certain species are restricted are very critical. This information is very critical for correlating land cover/use change and population trends for certain IBA trigger species and making conclusions on the trends in species populations based on species-habitat relationships.

Regarding conservation interventions (protection and long-term site conservation initiatives), both are less significant in terms of reversing current trends of land cover change. For example, in this study, protection does not seem to prevent deleterious land cover change while long-term engagement of SSGs has not translated into better conservation outcomes. Even though this may portray the true picture, there is need for further analyses taking into account other factors including perhaps a longer period working with SSGs as well as increasing the sample size of SSGs. From this study, it is evident that some of these conservation interventions take longer to demonstrate impact.

In view o these land cover change, it would be vital for future analyses to understand historical factors that could have accelerated habitat loss at key biodiversity sites. For example, Plumptre *et al.*,(2001), demonstrate how the 1994 genocide in Rwanda and the insecurity led to loss of species, complete extermination of two forest sites (Gishwati and Mukura forests) and a loss of Akagera National Park by 70%. Other studies provide in detail the impact of armed conflicts on biodiversity (Shambaugh *et al.*, 2001), even though other studies (Lindsell *et al.*, 2011) show peacetime pressures from clearance of forests for agriculture, logging and mining far exceed the pressures from civil war.

In conclusion, it is also important to understand the mechanisms leading to land use and land cover changes in the past as well as the impact of the past conservation interventions on biodiversity. Only then will it be feasible to understand the current trends and changes and predict future ones. It is also crucial that the investments made through working with communities be quantified for respective sites. Besides the short period the SSGs have been in operation, perhaps the results could have been different if the investments (resources/funding) made at each site had been monitored and included in the analyses. The level of conservation investment also could perhaps determine the level of impact demonstrable through less deleterious land cover change. The nature of activities that the respective communities are involved in all matters a lot. For example, if the SSG programmes at sites focus mainly on improving livelihoods at the expense of actual conservation, relative differences of conservation outcomes across sites will be expected.

CHAPTER FIVE: CONCLUSIONS, RECOMMENDATIONS AND CONTRIBUTION TO SCIENCE

5.1. Conclusions

The objectives of this study were:

i. to determine the percentage and extent of land cover change at IBAs using remotely sensed data.

ii. describe correlates of actual habitat change on these IBAs that could be used to inform monitoring strategies, conservation management and advocacy.

iii. based on the first two key objectives, predict IBAs that are at risk of deleterious land cover change

iv. to develop methodologies and protocols for use of remote sensing in IBA monitoring.

These objectives have been achieved and it is hereby concluded as follows:

- Habitats are in a state of permanent flux and rates and extent land cover vary from one site to another (spatially) and from one time period to another (temporally) and these changes are not uniformly distributed due to existing site specific factors. It is also evident that both land cover conversions and modifications taking place are human induced.
- Agricultural intensity as a correlate of change is the major factor influencing observed deleterious land cover change in and around key biodiversity sites
- Population density around key biodiversity sites did not show significant cause and effect on land cover change. However, there is need for some degree of caution when interpreting the non-significant attributes of population density on land cover change. This is because, there may be other underlying demographic factors that were not

considered in this study and probably if this had been done; the results could have been different.

- The two forms of site management and conservation interventions (designation of protected areas and BirdLife presence through site-based conservation and livelihood initiatives involving working with SSGs), did not demonstrate their impact in preventing deleterious land cover change.
- The above non-significant results on impact of protection and SSGs in averting deleterious land cover change need to be interpreted with a lot of caution as there may be other aspects that may require more investigation (e.g. management effectiveness of protected areas). For example, this would provide a better understanding as to why protection may not be effective in some places. Intrinsic, subtle and underlying factors operating at different scales or levels may have masked the importance of protection as an effective intervention in reducing land cover change at sites.
- Even though earth observation data are widely used to identify common land cover types (e.g. forest), scarcer land cover types and gradations may be more difficult to identify. For example, herbaceous and shrubland cover types were difficult to identify and interpret as compared to forest. This could be attributed to seasonality (dryness or wetness) and the seasonal variation in phenology causing associated variation in the spectral signatures for these land cover types thus making the image interpretation an arduous process.
- Cloud cover remains one of the major challenges in land cover analyses using satellite images and depending on the amount of cloud, some images become redundant when key details become obscured by clouds. However, with reducing costs, the use of images

acquired by passive or active microwave sensors, which are able to penetrate cloud cover will in the near future help to overcome the problem of cloudy days and satellite imagery.

- The cause and effect of land cover change is intricate and beyond what is covered in this study. Thus it may require substantial studies with as many related variables as possible included in the regression model.
- There may be other threats, for which real time monitoring would not be easy using satellite imagery (e.g. monitoring illegal activity such as logging).
- This study demonstrated a novel application of GIS and remote sensed data using a simple but effective remote sensing tool and hence very useful techniques for monitoring land cover change at key biodiversity sites.

5.2. Recommendations

5.2.1. Recommendation for action

The land cover analyses generated from remotely sensed data will be useless if not linked to research, land cover and land use policy, improved management of ecosystems and environmental education and public awareness efforts. The more the land cover change analyses products are translated into action, the more this technique will be adopted by many conservation practitioners. The recommendations from this study include:

- The need for translating of the results into policy and advocacy, management, conservation and monitoring of IBAs as well as other key biodiversity sites. At a policy and advocacy level, these results can be applied in many ways. For example, policy and advocacy strategies need to be scaled up through both bottom-up and top-down approaches. There is need to lobby the designations of sites that are still unprotected, improve policies associated with implementation of protected areas, scale up conservation efforts, formulate and promote environmental friendly agricultural and development policies.

- There is need to strengthen and increase management effectiveness of protected areas in order to benefit biodiversity and people (e.g. ecosystem services). Protection itself as a status is not a panacea to all conservation problems and neither does it translate into better conservation of biodiversity. Protected area agencies have to be supported in mobilising resources and capacity to maximise conservation efforts. Part of this strategy will mean increased collaboration by relevant government agencies with all stakeholders including the civil society, development agencies, local communities in mobilizing resources, planning, management, implementation of conservation programmes and monitoring of land cover and species. To achieve this, an integrated landscape planning approach that takes into consideration existing land uses, multifunctional landscapes and landscape configurations is critical if the dynamics and deleterious processes within the matrix around these sites are to be minimized.
- The results from this study have broader implications for conservation in terms of identifying key direct divers of land cover change and what needs to be done to avert further land cover change. This is very vital in the process of setting priorities to maximise conservation outcomes.
- The anthropogenic activities outside key biodiversity sites are critical when considering developing priorities for conservation or conservation planning. Even though there is need to keep track of these anthropogenic activities, it is incumbent upon conservationists to identify the types and locations of activities that are more deleterious and formulate appropriate mitigation measures aimed at reducing their impact on the integrity and

ecological functioning of the site. This will help in understanding the suite of threats that could be driving habitat loss.

- Management strategies within key biodiversity sites (e.g. IBAs), cannot be independent of human activities surrounding them in the multifunctional landscape. Considering that agricultural intensity around IBAs has been confirmed as one of the main drivers of land cover change at IBAs, measures should be put in place to reverse this trend and ensure the long-term survival of crucial habitats and species at IBAs. It is thus recommended that besides focusing conservation interventions at the site, there is need to work with communities and the relevant government agencies and key productive sectors (e.g. agriculture, forest, wildlife) and mainstreaming biodiversity conservation into these threat sectors to promote sustainable nature-based enterprises (e.g. agroforestry initiatives) and other environmentally friendly practices.
- Engaging the surrounding communities in a suite of activities ranging from conservation to promoting alternative sustainable nature-based livelihood enterprises that have a clear link between biodiversity conservation and poverty alleviation should be a priority intervention. This will promote biodiversity conservation in the surrounding matrix (e.g. agribiodiversity) and has the potential to improve livelihoods, ecological resiliency and provide the much needed ecosystem services. Subsequently, such novel approaches will improve people's livelihoods and relieve these sites of the pressure emanating from the overexploitation of natural resources within the sites.
- There is need for more synergies between the various sectors (e.g. agriculture, wildlife, forestry, health) that will ensure a more coordinated approach in planning and harmonising and mainstreaming conservation into development sectors.

- As far as working with communities (SSGs) is concerned, there is need to develop baseline data (e.g. biodiversity and socio-economic) when SSGs are established so that this can form a baseline for future efforts to track and evaluate the impact of their conservation work. Even though socio-economic data was not part of this study, it is also recommended that such data be collected continuously to aid future monitoring of the impact of conservation and livelihood enterprises on both biodiversity and poverty alleviation.
- For the SSG approach to be more effective there is need to put emphasis on the links between biodiversity conservation and poverty eradication. Without taking this into consideration and demonstrating this link, this SSG approach may end up being skewed towards promoting livelihoods at the expense of biodiversity conservation.
- Since it is apparent from this study that SSGs are not preventing deleterious land cover change outside the IBAs, it is recommended that the approach needs review in the context of the need to also focus not only on the IBAs themselves but also outside IBAs as part of conservation at a landscape level.
- Awareness raising efforts should be scaled up. Empowerment of the local people and capacity building should include environmental education particularly to spearhead behavioural change. If this can start at a site level, it will have a ripple effect at other levels and achieve greater impact and develop a larger constituency of people who are passionate about biodiversity conservation.
- To address the problem of existing data gaps and paucity of information including georeferenced bird data, systematic monitoring data, infrastructure data layers among others, there is need to develop baseline data (georeferenced habitat and species data) as

benchmarks for future monitoring. The conservation fraternity should place greater emphasis on the collection of such information and development of databases.

- Even though remote sensing is a very useful tool in monitoring key biodiversity sites, ground-truthing also helps in reducing such errors so that for example, a closed prolific stand of an alien invasive species (e.g. *Prosopis juliflora*) or even closed mango plantation is not mistaken for a natural closed forest important for fauna.
- Need to develop and manage spectral libraries which provide the signatures of various rangeland condition classes and land cover categories will in future reduce problems with land cover classification. And in so doing even reduce costs of ground truthing.
-
- From a conservation point of view, the results from this study require quite a number of substantial considerations on the way relevant government agencies and conservation organisations need to work in order to reverse the current trend in land cover change at key biodiversity sites.

5.2.2. Recommendations for further studies

Considering the results from this study, this thesis work has generated some specific and broader research questions that can be addressed by future studies. The main key recommendations identified from this study include:

- The need to investigate further and identify other threats that may be operating singly or interacting in synergies. For example, forest and grassland habitats are more vulnerable when the extents of agriculture intensity and population density work in synergy.
- Correlate changes in land cover with bird diversity, temporal and spatial changes in species population and geographical distribution at IBAs. For example, future land cover

change studies should based on availability of historical species data correlate the impact of land cover change on populations of focal flagship species or species assemblages at these sites.

- Sustain land cover change analyses and patterns of land cover change and parameters that drive these changes used in modelling future trajectories of land cover change. Results from modelling will make it possible to mitigate and initiate timely preventive interventions that would check future loss of biodiversity. This is even more necessary considering that modelled predictions for 2100 reveal that the largest impact on biodiversity (e.g. tropics) is expected to be due to land use and land cover. Having these models will be useful in considering precautionary principle and preventive strategies in combating deleterious land cover change.
- Investigate in historical context what events triggered the observed land cover change (both positive and negative) during the respective period (e.g. fire, pests, invasive species, volcanic eruptions, wars, extreme droughts). Land-cover change is a discrete process and characterised by periods of rapid change often triggered by an events at a local, national, regional or global levels. Future studies should aim at
- Investigate whether one threat can also initiate or trigger a cascade of changes (modification and conversion) in land cover within a site.
- Research in detail how past government policies accelerated or inhibited detrimental land cover change at national level with a view of developing a whole set of factors driving land cover change. Examples of such policies or reform processes may include logging bans, directives on forest excisions, degazzettement of sites, changes in land tenure systems, encroachment, agricultural policies, and agricultural incentives.

- There are other potential factors causing environmental change in Africa that would need to be factored in subsequent correlations include population growth, biophysical factors, economic pressures and policies, food production systems, mining, industry and energy
- Correlate land cover change with infrastructural developments in the transport sectors (e.g. road density and location) and what impact road networks criss-crossing key biodiversity sites would have on population density in and around these sites and thereafter on land cover change) and urbanization (e.g. how proximity to urban centres influence the rates of land cover change).
- Correlate rates of change with other biophysical factors such as altitude, steepness (slope and topography for forest sites and fore regime, drought in rangeland sites), ruggedness and inaccessibility of the sites to human activities to better understand the mechanics of why deforestation is inhibited or accelerated in those areas.
- As for protection, there is need for more investigation on other impacts of anthropogenic factors outside protected areas. This may include looking at the impacts of human activity on invasive species, poaching, hunting and fire outbreaks and other latent threats. Further studies should investigate if there sufficient evidence to link observed differences in land cover change in protected areas to leakage. As for protected areas; additional studies may assess whether land cover change varies with distance to protected area boundary or if the duration since a protected area was designated is critical factor in preventing deleterious land cover change. These additional analyses may provide useful insights into other underlying factors that either impede or accelerate the rate of land cover change.

- In terms of land cover classification, there may be cases of interpretation errors of certain land cover classes owing to land-cover seasonality and variability (e.g. based on the period) and near permanent cloud cover in certain parts of Africa; these are likely to introduce interpretation errors. To overcome this problem, additional land cover analyses may need to use cloud penetrating radar imagery provided by such satellite platforms as the Japanese Earth Resources Satellite (JERS-1) and the European Remote Sensing Satellite (ERS) as an alternative to study tropical forest cover.
- Finally, remote sensing techniques will become more efficient and effective if there is combined use of moderate, high and very high spatial resolution data with the latter being useful when there is a need to zero in onto the more sensitive and priority sites where the deleterious rates of land cover change have been noted to be higher. This will provide more details about the extent and actual diverse drivers of land cover change at site level.

5.3. Contribution to Science

This is the first rapid region-wide assessment of land cover and correlates of land cover change within Important Bird Areas of Eastern Africa using remotely sensed data and a pioneer study to use the simple but cost-effective Land Cover Change Graphical User Interface Tool (LCC-GUI) for image interpretation and analysis. The study therefore demonstrates the capability of LSS-GUI as a prototype tool that can be applied in land cover analyses at IBAs elsewhere in the world.

The study provides qualitative and quantitative analyses of LCC within and outside key biodiversity sites. This would have been impossible using field-based surveys only. Most land cover change analyses has focused on single sites and undertook analyses within the sites only. However, considering that land cover and land use dynamics outside key biodiversity sites have a huge impact on biodiversity within the site, the results from this work will be instrumental in reviewing past conservation interventions (protection and SSG approach) as well as guide future work within and outside IBAs.

This is also the first study to use remotely sensed data to review and evaluate the impact of the BirdLife International's SSG conservation approach in reducing land cover change at IBAs since the SSG model was adopted by BirdLife International. This study will stimulate discussions on why it is important to take into account many factors when developing programmes and undertaking conservation planning as well as setting priorities to reverse land cover change and biodiversity loss. For example, the impact of changes, threats and pressures emanating from anthropogenic activities from the surrounding landscape on biodiversity at the key biodiversity sites and the dynamics within the matrix can lead to deleterious land cover change. Based on land cover, ongoing or past land cover change and the correlates of land cover, it is possible as early warning systems to identify IBAs that are vulnerable for listing amongst the IBAs in danger and where most urgent interventions and allocation of resources is needed to stem the current rates of land cover loss. The findings from this study have important implications for the conservation of species, sites and habitats that are severely affected by deleterious land cover change.

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ANNEXES

Annex 1. Global IBA identification Criteria

A1. Globally threatened species

Criterion: The site is known or thought regularly to hold significant numbers of a globally threatened species, or other species of global conservation concern.

Notes:

The site qualifies if it is known, estimated or thought to hold a population of a species categorised by the IUCN Red List as Critically Endangered, Endangered or Vulnerable. In general, the regular presence of a Critical or Endangered species, irrespective of population size, at a site may be sufficient for a site to qualify as an IBA. For Vulnerable species, the presence of more than threshold numbers at a site is necessary to trigger selection. Thresholds are set regionally, often on a species by species basis. The site may also qualify if holds more than threshold numbers of other species of global conservation concern in the Near Threatened, Data Deficient and, formerly, in the no-longer recognised Conservation Dependent categories. Again, thresholds are set regionally.

A2. Restricted-range species

Criterion: The site is known or thought to hold a significant component of a group of species whose breeding distributions define an Endemic Bird Area (EBA) or Secondary Area (SA).

Notes:

This category is for species of Endemic Bird Areas (EBAs). EBAs are defined as places where two or more species of restricted range, i.e. with world distributions of less than 50,000 km^2, occur together. More than 70% of such species are also globally threatened. Also included here are species of Secondary Areas. A Secondary Area (SA) supports one or more restricted-range species, but does not qualify as an EBA because less than two species are entirely confined to it. Typical SAs include single restricted-range species which do not overlap in distribution with any other such species, and places where there are widely disjunct records of one or more restricted-range species, which are clearly geographically separate from any of the EBAs.

A3. Biome-restricted species

Criterion: The site is known or thought to hold a significant component of the group of species whose distributions are largely or wholly confined to one biome.

Notes:

This category applies to groups of species with largely shared distributions of greater than 50,000 km^2, which occur mostly or wholly within all or part of a particular biome and are, therefore, of global importance. As with EBAs, it is necessary that a network of sites be chosen to protect adequately all species confined to each biome and, as necessary, in each range state in which the biome occurs. The 'significant component' term in the Criterion is intended to avoid selecting sites solely on the presence of one or more biome-restricted species that are common and adaptable within the EBA and, therefore, occur at other chosen sites. Additional sites may, however, be chosen for the presence of one or a few species which would, e.g. for reasons of particular habitat requirements, be otherwise under-represented.

A4. Congregations

Criteria:

A site may qualify on any one or more of the four criteria listed below:

i). Site known or thought to hold, on a regular basis, 1% of a biogeographic population of a congregatory waterbird species.

ii). Site known or thought to hold, on a regular basis, 1% of the global population of a congregatory seabird or terrestrial species.

iii). Site known or thought to hold, on a regular basis, 20,000 waterbirds or ³ 10,000 pairs of seabirds of one or more species.

iv) Site known or thought to exceed thresholds set for migratory species at bottleneck sites.

Annex 2. List of Study Sites

No.	IBA Code	International IBA Name	Country	SitArea (Ha)	SitLong	SitLat
1.	BI002	Kibira National Park	Burundi	37870	29.3	-2.7
2.	BI003	Ruvubu National Park	Burundi	43630	30.4	-3.1
3.	BI004	Rusizi National Park	Burundi	9000	29.23333	-3.2
4.	BI005	Bururi Forest Nature Reserve	Burundi	3300	29.58333	-3.93333
5.	ET023	Entoto Natural Park and escarpment	Ethiopia	13000	38.75	9.183333
6.	ET029	Akaki-Aba-Samuel wetlands	Ethiopia	12068	38.06667	8.866667
7.	ET039	Koka dam and Lake Gelila	Ethiopia	18400	39	8.5
8.	ET041	Lake Zeway	Ethiopia	65400	38.81667	7.983333
9.	ET047	Lake Langano	Ethiopia	65400	38.76667	7.616667
10.	ET048	Abijatta--Shalla Lakes National Park	Ethiopia	88700	38.5	7.5
11.	ET054	Bale Mountains National Park	Ethiopia	247000	39.71667	6.75
12.	ET064	Yabello Sanctuary	Ethiopia	250000	38.41667	4.916667
13.	KE001	Aberdare Mountains	Kenya	179900	36.66667	-0.41667
14.	KE002	Kianyaga valleys	Kenya	12000	37.33333	-0.5
15.	KE003	Kikuyu Escarpment forest	Kenya	37600	36.66667	-0.93333
16.	KE004	Kinangop grasslands	Kenya	77000	36.56667	-0.7
17.	KE005	Mount Kenya	Kenya	271000	37.33333	-0.16667
18.	KE006	Mukurweini valleys	Kenya	30000	37.11667	-0.5
19.	KE026	Chyulu Hills forests	Kenya	18000	37.83333	-2.58333
20.	KE029	Machakos valleys	Kenya	5000	37.23333	-1.58333
21.	KE031	Meru National Park	Kenya	87000	38.41667	-0.3
22.	KE033	Samburu and Buffalo Springs National Reserves	Kenya	29600	37.5	0.666667
23.	KE036	Nairobi National Park	Kenya	11700	36.96667	-1.3
24.	KE037	Dunga swamp	Kenya	100	34.78333	-0.16667
25.	KE038	Koguta swamp	Kenya	200	34.76667	-0.28333
26.	KE039	Kusa swamp	Kenya	350	34.85	-0.31667
27.	KE041	Yala swamp complex	Kenya	8000	34.18333	0.083333
28.	KE042	Amboseli National Park	Kenya	39200	37.1	-2.55
29.	KE043	Cherangani Hills	Kenya	95600	35.85	1.266667
30.	KE047	Lake Magadi	Kenya	10500	36.28333	-1.86667
31.	KE048	Lake Naivasha	Kenya	23600	36.35	-0.76667
32.	KE051	Mau forest complex	Kenya	273300	35.33333	-0.5

33.	KE052	Mau Narok-Molo grasslands	Kenya	40000	35.91667	-0.55
34.	KE055	South Nandi Forest	Kenya	18000	35	-0.08333
35.	KE057	Busia grasslands	Kenya	250	34.25	0.416667
36.	KE058	Kakamega forest	Kenya	18300	34.88333	0.283333
37.	KE059	Mount Elgon	Kenya	95000	34.63333	1.333333
38.	RW003	Akagera National Park	Rwanda	100000	30.63333	-1.75
39.	RW007	Nyungwe forest	Rwanda	90000	29.23333	-2.5
40.	TZ003	Mount Kilimanjaro	Tanzania	166100	37.33333	-3.08333
41.	TZ005	Mahali Mountain National Park	Tanzania	323000	29.83333	-6.2
42.	TZ008	Rubondo Island National Park	Tanzania	45700	31.83333	-2.33333
43.	TZ014	Burigi-Biharamulo Game Reserves	Tanzania	350000	31.25	-2.25
44.	TZ024	Kagera swamps	Tanzania	111600	30.83333	-1.5
45.	TZ026	Lake Kitangire	Tanzania	12000	34.3	-4.1
46.	TZ029	Mtera reservoir	Tanzania	66000	35.83333	-7.08333
47.	TZ030	Nyumba ya Mungu reservoir	Tanzania	22000	37.33333	-3.66667
48.	TZ031	Lake Natron and Engaruka basin	Tanzania	154000	36	-2.41667
49.	TZ032	Rufiji Delta	Tanzania	72000	39.45	-8
50.	TZ034	Singida lakes	Tanzania	1100	34.7	-4.3
51.	TZ046	Bagamoyo District coastal forests	Tanzania	20000	38.66667	-6.16667
52.	TZ062	North Pare Mountains	Tanzania	3000	37.66667	-3.75
53.	TZ063	South Pare Mountains	Tanzania	25000	37.83333	-4.33333
54.	TZ064	Rubeho Mountains	Tanzania	62861	36.53333	-7
55.	TZ065	Mount Rungwe	Tanzania	31542	33.66667	-9.13333
56.	TZ071	West Usambara Mountains	Tanzania	38169	38.33333	-4.66667
57.	TZ074	Longido Game Controlled Area	Tanzania	280000	36.83333	-2.83333
58.	UG002	Echuya Forest Reserve	Uganda	4000	29.81667	-1.28333
59.	UG004	Bwindi Impenetrable National Park	Uganda	33100	29.66667	-1.05
60.	UG006	Kibale National Park	Uganda	76600	30.33333	0.433333
61.	UG010	Semliki reserves	Uganda	115000	30.33333	1
62.	UG012	Mabira Forest Reserve	Uganda	30600	33	0.5
63.	UG015	Lutoboka point (Ssese islands)	Uganda	200	32.28333	-0.3
64.	UG016	Nabugabo wetland	Uganda	22500	31.91667	-0.45
65.	UG017	Mabamba Bay	Uganda	16500	32.33333	0.083333
66.	UG018	Lutembe Bay	Uganda	800	32.56667	0.166667

67.	UG019	Budongo Forest Reserve	Uganda	79300	31.58333	1.75
68.	UG021	Ajai Wildlife Reserve	Uganda	15800	31.16667	2.866667
69.	UG027	Lake Opeta	Uganda	56600	34.16667	1.666667
70.	UG028	Mount Elgon National Park	Uganda	114500	34.5	1.166667
71.	UG029	Mount Moroto Forest Reserve	Uganda	48300	34.7	2.7