LAND USE CHANGE IN MAASAILAND

DRIVERS, DYNAMICS AND IMPACTS ON LARGE-HERBIVORES AND AGRO-PASTORALISM

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Thesis Certification

I declare that the work in this dissertation was carried out in accordance with the regulations of the University of Edinburgh, is wholly my own work unless otherwise stipulated, referenced and/or acknowledged in the thesis. This dissertation has not been submitted for qualifications at any other academic institution.

Fortunata Urban Msoffe

SIGNED………………………..… DATE… 31-08-2010 …

Abstract

The Maasailand of Kenya and Tanzania supports one of the richest wildlife populations remaining on Earth. However, over the last century, Maasailand has experienced land transformation notably through conversion of former rangelands to croplands. With the anticipated human population increase in East Africa, more impacts should be envisaged on these rangelands.

This thesis investigates the root causes and underlying drivers of land-use change in the Maasai-Steppe ecosystems, stemming from historical, socio-cultural, political as well as the biophysical conditions. To analyse the different drivers of change, an integrated methodological approach was employed. This included a collation of historical data and information derived from both gray and published literature, analysis of remote sensing and Geographical Information Systems (GIS) data, field surveys, workshops, observations, as well as personal communications.

Observed land-use change from savannah rangelands to expansive croplands are mainly linked to government policies, land tenure, human population growth (which is also likely to be the largest future driver) and climatic conditions. Consequently these changes have impacted the agro-pastoralist community, whose main incomes for their livelihoods depend on pastoralism. Subsequent loss of formerly communal grazing lands to establish protected areas; large-scale farming and/or private ranches have aggravated the problems of sedentarization due to villagization and privatization policies of the formally mobile agro-pastoral communities.

Land-use change also had negative impacts on migratory wildlife species, particularly those utilizing both protected areas and dispersal ranges in communal and/or private lands. The impacts ranged from loss of their migratory routes and corridors to massive declines of populations due to the loss of access to grazing resources. The study recommends government’s interventions for keeping the land open for access to grazing resources as well as opening up wildlife corridors, where it is deemed necessary for national interests.

1 Chapter One: General Introduction

1.1 Background

Over the last two to three decades there has been a notable change in land-uses in the Maasai-Steppe ecosystems, especially from subsistence to extensive agriculture (Peterson 1978,Ecosystems Ltd. 1980,Borner 1985). This has led to a growing concern about the sustainability of the system in supporting the rich biodiversity and the agro-pastoral livelihoods (Mwalyosi 1995,Kahurananga & Silkilwasha 1997,OIKOS 2002). Some of these changes have been influenced by political factors and the linkages between policies and ecological changes are poorly documented as noted also in other Maasai land in Kenya (Campbell *et al.* 2005). Notable conversions to agriculture by pastoralists in the rangelands have been linked partially to issues of land tenure, insecurity and livelihood needs particularly of the poor and the most vulnerable families (Homewood *et al.* 2001,Sachedina 2008). These factors and others have impacted negatively on the population of large herbivore and also the livelihoods of the local Maasai community, Msoffe, *et. al.* (in press).

Despite the nation-wide strategic economic value of wildlife in the Maasai-Steppe ecosystems, scientific evidence illustrates that wildlife numbers are declining at an unsustainable rate (TAWIRI 2001), while human residents within the ecosystem are dealing with increasing rates of poverty (Muir 1994,TNRF 2008). Large mammal numbers within the ecosystems have declined precipitously over the past decade; zebra (*Equus burchelli*) and wildebeest (*Connochaetes taurinus*) populations have been reduced to fractions of their former levels and species like oryx (*Oryx beisa*) and kongoni (*Alcelaphus buselaphus*) are now at threat of local extinction (TWCM 2000). The causes of these declines are thought to be linked with habitat loss in the dispersal areas and calving grounds and poaching (TAWIRI 2001, OIKOS 2002). While the economic potential of conservation activities remains relatively untapped by the Maasai communities, maintaining safe dispersal grounds for large animals migrating within the area is crucial to ensure the future ecological and economic health of this system. This thesis will undertake to first understand what drives these changes, secondly through use of remotely sensed and local knowledge establish the changes (land cover changes, (Wessels *et al.* 2004) and finally undertake a statistical exercise that tries to link the land changes to the biophysical landscape variables (Serneels &

Lambin 2001c,Hunter *et al.* 2003) as well as establishing the impacts and consequences on wildlife and agro-pastoralism.

1.2 Drivers and causes of land use and land cover change

Changes in terrestrial ecosystems brought about by human activity are driven by land cover conversion, land degradation, land use intensification or other forms of land modification (Mertens & Lambin 2000). The term 'land cover' refers to the attributes of a part of the Earth's land surface and immediate subsurface, including biota, soil, topography, surface and ground water and human structures. On the other hand the term 'land-use' refers to the purpose for which humans exploit the land cover (Lambin 2000). Changes in human land use are frequent causes of land cover conversion (the complete replacement of one cover type) and modification (more subtle changes that affect the character of land cover without its overall classification). Moreover, land-use change is a major driver of habitat modification and can have important implications for the distribution of species and therefore for entire ecological systems (Serneels & Lambin 2001b).

Land-use changes are cumulatively transforming land cover at an accelerating pace, mainly in the tropics (Lambin *et al.* 2000). These changes in terrestrial ecosystems are closely linked with the issue of the sustainability of socio-economic development since they affect essential parts of our natural capital such as climate, soils, vegetation, water resources and biodiversity (Turner II *et al.* 2007). Research on the causes of land-cover change from global to regional level indicated at global level three primary influences, which are; population, level of affluence and level of technology, while at regional scales rural-to-urban migration, economic growth, changes in lifestyle, and changing economic and political arrangements are the main drivers of change (Lambin *et al.* 2001). Other explanatory factors include the role of institutions and the influence of local culture (McCusker 2004). Processes related to land-cover conversions are complex and depend on the scale of analysis which can be linked to economic, cultural, political, institutional and demographic factors (Campbell *et al.* 2005). Recent research work on processes related to land cover conversions has taken an integrated approach in trying to link causes and processes. Analysis of land conversions at multiple scales demands conceptual frameworks and analytical methods that are both comprehensive enough to capture the dynamics of society-environment interactions at different scales and flexible enough to accommodate the temporal dynamics of these processes (Campbell *et al.* 2005). Determining the effects of land-use and land-cover change on the Earth system depends on an understanding of past land-use practices, current land-use and land-cover patterns, and projections of future land use and cover, as affected by human activities, population size and distribution, economic development, technology, markets, climate and other factors. The impacts of socio-political and land cover changes through land conversion, modification and fragmentation on biodiversity is poorly understood (Campbell *et al.* 2005).

In East African savannas people, wildlife and livestock have co-existed for millennia (Thornton *et al.* 2002). However this has declined since 1950's as conservation policies have excluded people and livestock co-existence from newly created parks. Growing human populations and expanding agricultural activities around these parks have led to the declining wildlife populations and increasing people-wildlife conflicts (Gamassa 1988,Lama 1998,Carlsson 2004,Gadd 2005). Large areas of pastoral lands are now becoming fragmented with a large portion being converted into agricultural land, and thus increasing the exclusion of pastoralists and wildlife from land with the highest potential (Campbell *et al.* 2000). Land use policy is a major factor influencing the conversion of rangelands to cultivation (Homewood *et al.* 2001,Serneels & Lambin 2001a). Policy instruments in particular influence the decision-making process of agro pastoralists and therefore modify land use changes and their impacts on the ecosystem (Homewood *et al.* 2001,Homewood 2004).

1.2.1 Agriculture and land-use change

Agriculture has been the greatest force of land transformation on this planet. Nearly a third of the Earth's land surface is currently being used for growing crops or grazing cattle (Lambin & Geist 2006). Much of this agricultural land has been converted from natural forests, grasslands, and wetlands that provide valuable habitats for species and valuable ecosystem services for humankind (Millenium Ecosysem Assessment 2003). Much of the expansion of croplands came at the expense of forests, while much of today's grazing land was formerly natural grasslands; although there are notable exceptions to these trends, for example, the North American Prairies were lost to croplands, and many Latin American forests have been cleared for ranching.

Subsequently, the global forest area has decreased from ~53 million square kilometres in 1700 to ~43-44 million square kilometres today, while the area of savannas and grasslands has decreased from 30-32 million square kilometres to 12-23 million square kilometres (Lambin & Geist 2006).

The expansion of agriculture has shifted spatially over time, following the general development of human settlements and the global economic activity (Lambin & Geist 2006). Much of the large-scale cultivation in 1700 was concentrated in the Old World, specifically in Europe, the Indo-Gangetic Plains, eastern China, and Africa. Roughly 2-3% of the global land surface was cultivated at that time. Since then the rate of cropland expansion increased with European colonization and increasing globalization of world markets. New settlement frontiers were established in North America, Latin America, South Africa and the Former Soviet Union. North America and the Former Soviet Union experienced their most rapid expansion of cultivated land starting around 1850. In Latin America, Africa, and South and Southeast Asia Areas experienced slow cropland expansion until the 20^{th} century, but have seen exponential increases in the last 50 years. China had a steady expansion of croplands throughout most of the last three centuries (Lambin & Geist 2006).

1.2.2 Human population growth and land-use change

As a driving force of environmental change, human population growth is unique in its plausibility and ease of quantification (Meyer & Turner 1992). World population is expected to soar by 34% to reach 9.1 billion by 2050, with the entire 2.3 billion increase to take place in the developing countries (ESA-UN 2009). In the neo-Malthusian position, global population increases are accorded primary importance in most environmental change because of the resources required to sustain the demands of billions of people (Meyer & Turner 1992). Increasing people leads to greater pressure on the land and ultimate reduction in the ability of rangelands to support livestock and people. Effectively, increasing human populations intensify the demand for the rangeland resources beyond the ability of the land to provide them (Talbot 1986).

1.2.3 Impacts of land-use/cover change on biodiversity and ecosystems

Ecosystems provide regulating as well as supporting services that are essential for agriculture and fisheries. These include provisioning of food, fibre and water;

regulating services such as air, water and climate regulation, pollination and pest control; and providing resilience against natural disasters and hazards (Tilman 1999). The United Nations Environment Programmes Global Biodiversity Assessment estimates current extinction rates at 50 to 100 times "normal", and anticipates a tenfold or even 100-fold increase over the next quarter century, when between 2 and 25 percent of species could be lost (Millenium Ecosysem Assessment 2005). The primary cause of this loss is not hunting or overexploitation, though these play a part, but loss of natural habitat. Habitat loss is generally greatest where human population density is highest.

1.2.4 Land use/cover change and impacts on migratory species

Around the world, many of the most spectacular wildlife migrations have either disappeared due to human activities or are in steep decline (Wilcove & Wikelski 2008). Increasing pressure on land, through human population growth and associated agricultural expansion in the rangelands have resulted in increasing rates of loss of habitats for migratory species including blockage of their migration routes, declining wildlife populations (TAWIRI 2001) and the increasing insularization of protected areas with subsequent local extinction of species (Newmark 1996). In the North American Great Plains, hundreds of thousand of bison trekking across the prairies, have disappeared in a period of less than two centuries (Wilcove & Wikelski 2008).

1.3 The Maasailand- people, land use and changes

The Maa-speaking communities have dominated the pastoral niche in East African rangelands for centuries and they have been the most prominent and powerful communities prior the mid 19th century (Homewood & Rogers 1991). Traditionally, they are semi-nomadic pastoralists, with a very high dependency on livestock although they have also been practicing agriculture such as cropping on a subsistence scale (Borjeson *et al.* 2008). Maasai pastoralism declined towards the end of the twentieth century because of the constraints on their nomadic life styles and the development of alternative economic opportunities off the land (Groom 2007).

During the pre-colonial period, Maasai dominated lands were more largely managed as common property with access governed through social networks of sections, location, clan, kin and peer group friendship (Homewood *et al.* 2009). During the colonial period, large areas of Maasailand in Kenya and Tanzania were transferred to settlers (large-scale farmers) and protected areas establishment. The Southern Maasai Reserve (Fig. 1.1) which extended from Southern Kenya to Northern Tanzania, was designated as Trustland with access on the criteria of Maasai ethnicity, but soon became subject to intense pressure for access by origin from other ethnic groups whose main livelihood relied on crop-cultivation (Homewood *et al.* 2009). The area is increasingly home to non-Maasai, land use and land tenure have also been changing as well as increasingly diversified into other ways of livelihoods (Sachedina 2008).

Ecologically, Maasailand includes some of the most important tourist destinations in Africa, such as the Serengeti-Mara ecosystem, hosting abundant of wildlife (including the spectacular migrations of ungulates) for game viewing, sports and trophy hunting. For millennia, pastoralists have shared landscapes with wildlife throughout Africa (Homewood & Rogers 1991,Coast 2002). Throughout the 20th century, this co-existence has been in decline as conservation policy excluded people and livestock from protected areas, and expanding agriculture excluded wildlife and livestock use (Voeten 1999).

In this thesis, an integrated approach or conceptual framework (Fig. 1.2) is used in order to critically address some of the hidden underlying factors yet important drivers of land use/cover change. The approach will be used to facilitate the linkage of the quantitative and qualitative data necessary in understanding the problems and issues linked to land use, land tenure and wildlife conservation policies in relation to the agro-pastoral livelihoods in the Maasailand. More importantly this is one of the first studies to use an integrated approach in analyzing the drivers of land use/cover change and their impacts in extensive pastoral lands.

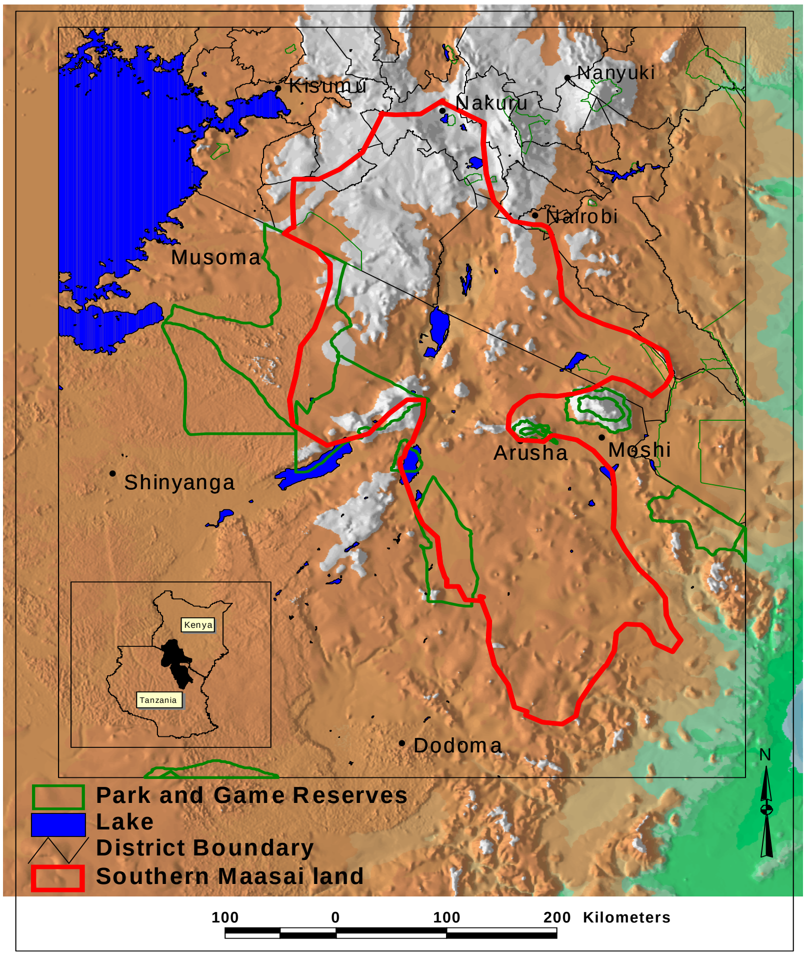

Figure 1.1 Map derived from Digital Elevation Model (DEM), showing the Southern Maasailand approximate boundary (thick red line) and protected areas (thick green lines) in East African rangelands (Said, *et al.* Unpubl.).

1.4 Studies on land use/cover change in Maasailand

Past research in Maasailand has led to the development of relative simple conceptual models to help analyze the critical pressure points and thresholds in changing land use and wildlife populations, for example, (Homewood *et al.* 2001,Serneels & Lambin 2001c,Homewood *et al.* 2009). These studies however, pointed out the need to further understand the following questions that will be partly addressed in the current study:

- What are the main determinants shaping livelihoods and triggering change in the study area?
- To what extent do external factors such as biophysical and eco-climatic/agro-ecological factors on the one hand, and infrastructure and policy on the other, shape livelihood choices?
- To what extent are livelihoods determined by socio-demographic characteristics of the household?
- What trends do these patterns indicate in terms of land use change, poverty trajectories and wildlife conservation?

Scale	Question
Regional	Do wildlife and land use trends differ across savanna ecosystems of the Maasai land in Kenya and Tanzania? Do trends differ inside and outside reserves?
Landscape	• Understanding the dynamics of land cover change at various spatial scales? • What are the mechanisms behind the interaction of people (cropland, settlement, conservation areas, livestock grazing) and wildlife at fine scales? • How is land use changing and how has this affected the number and distribution of wildlife populations, wildlife habitat and important migratory routes at the ecosystem scale?
Local (across pastoral to protected area boundaries)	• How do different forms of land use affect forage and water availability for wildlife and other species?

Increasing perception of broad pattern (what, where)

Increasing understanding of process (how, why)

Figure 1.2 Conceptual framework for the study, adapted from Reid *et al.* (Unpub.)

1.5 Research Aims

Figure 1.2 presents the framework of the study in terms of trying to discern broad patterns of changes but also capture the process so as to understand further the mechanisms of these changes. This PhD research project was part of a regional study undertaken by the International Livestock Research Institute (ILRI) to analyze broad patterns of trends in wildlife and land use across savanna ecosystems of the Maasai-land in Kenya and Tanzania (Fig. 1.2). This thesis focused at the regional, landscape and local scale levels and reflected more on understanding the processes (the how and why) that influence the dynamics and mechanisms of land cover conversion and land use change, and the consequences for, and impacts on large wild herbivore in the ecosystem. Further, this work attempted to address at the local scale how different forms of land use affect forage and water availability for migratory wildlife and other species.

1.5.1 Thesis objectives

1. To analyse the broad historical changes in land use, land tenure, socio-cultural, human demography, policies and legislation and how they have affected trends in wildlife and livestock population, key resources and habitats using an ecological-socio-political framework.

2. To analyse the drivers and causes of land use change in the Maasai-Steppe of Northern Tanzania using multiple logistic regressions and spatial statistics

3. To assess the distribution of wildlife, livestock and people in communal lands of the Tarangire-Simanjiro ecosystem during the wet season using a participatory field survey approach

4. To analyze and synthesize the status of meta-population of a key migratory wildlife species from key ecosystems covering protected and un-protected lands of East African savanna rangelands

1.6 Thesis Structure

The thesis consists of four main data Chapters 2-5, which are self contained and understandable on their own. These chapters have either been published (chapter 4) accepted for publication (Chapter 2), under review for re-submission (Chapter 3) and/or in preparation for submission to publication (Chapter 5).

Chapter 2: This chapter explores the underlying drivers and causes of land-use/cover change in the Maasai-Steppe of Northern Tanzania, using an ecological-socio-political analysis framework. The chapter addresses the hypothesis that, while there are a number of socio-cultural, political and historical factors driving land-use change in the ecosystem, rangeland conversions to agriculture over the years has continued to impact negatively the large-migratory wildlife species and agro-pastoralism. These changes have in turn jeopardized the ecological integrity of the landscape.

Chapter 3: This chapter continues to look at the drivers of land-use change by using an empirical modelling approach. The chapter tests the relationship between land use change and the biophysical variables in the landscape. The findings suggest that future land cover conversions to agriculture would be constrained by the values of the biophysical variables from the global model in this study.

Chapter 4: This chapter demonstrates a wildlife monitoring survey in a communal land where majority of the key migratory species disperse into during the wet season and intermingle with livestock and people. As key dispersal and calving grounds for the migratory species this area needs to be protected by the community, whom in this case have to be empowered and involved in the process. The study shows that local communities appreciate the value of wildlife conservation more if they are part-and parcel of the monitoring system in place.

Chapter 5: This chapter reviews and describe the status of a key migratory species meta-population from five ecosystems in the East African rangelands based on a synthesis of information from previous studies. The chapter further discusses the processes and drivers which have lead to loss of its habitat. The findings from this study highlights the urgent need for interventions in order to conserve the migratory patterns that covers both protected and unprotected lands.

Chapter 6: This chapter summarizes the main findings from the study and then discusses implications of its results within the global context, their relevance within the discipline of land use change science and biodiversity conservation at the landscape/ecosystems levels.

2 Chapter Two: Drivers and impacts of land-use change in the Maasai Steppe of Northern Tanzania: an ecological-social-political analysis[i]

Abstract

This chapter discusses the drivers and impacts of land-use change at a regional level in the Maasai-Steppe, Northern Tanzania. An ecological-socio-political analysis approach was adapted to unfold and synthesize the causes of land-use change emanating from historical, political and livelihood needs. Remote sensing data were used to analyze land use change and GIS was used to link-up with wildlife and livestock population dynamics and distribution data derived from aerial censuses.

Results indicated that agricultural land increased five-fold within the study period, while human population increased exponentially from 3.3% pa in 1988 to 3.4 % pa in 2002. On the other hand, wildlife migratory routes used by key species declined from 9 in 1964 to 5 in 2000, out of which 3 were seriously threatened by blockage through extensive cultivation spreading in the study area. Recurrent droughts and diseases have contributed to the declining livestock economy over the years due to livestock loss and the unpredictable and erratic rainfall has limited their recovery. Efforts to reverse the on-going trends should include community-based wildlife ventures supported by proper land-use plans in order to generate direct tangible benefits from wildlife to communities while maintaining the ecosystem viability.

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[i] Materials from this chapter have been accepted for publication as: Msoffe, F.U., Kifugo, S.C., Said, M.Y., Neselle, M., van Gardingen, P., Reid, R.S., Ogutu, J.O., Herero, M. and de Leeuw, J. (*In Press*) Drivers and impacts of land-use change in the Maasai Steppe of Northern Tanzania: an ecological-social-political analysis. *Journal of Land Use Science*

My contribution in this paper: In collaboration with my supervisory team (Reid, van Gardingen, Said, Ogutu, Herero and de Leeuw) I developed the concept and worked on it, collated all the data from field, library, and archives as well as from various institutions and presented this concept in a workshop that was organized by ILRI and AWF in the study area. I then did the analysis of remote sensing data with the support of Kifugo under the close supervision of Said. Kifugo, Neselle and I did the ground truthing and interviews with local Maasai in the study area. Ogutu advised and helped on the statistical analyses and read several versions of draft manuscripts along with my supervisory team. I finally worked on the draft manuscript and de Leeuw provided critical comments in shaping the final manuscript that was submitted and accepted for publication.

2.1 Introduction

Land use and land cover change (LULCC) is a central component of global environmental change with direct implications for the Earth's climate, ecology, and human societies, and is of great concern to national and international policymakers (Campbell *et al.* 2005). Policymakers seek from scientists information on the root causes of LULCC in order that policy may focus not on symptoms, but upon the fundamental processes that require remedial action. However, processes that drive LULCC are complex and require the use of multiple methods of analysis and critical interpretation of social data in order to understand the drivers and impacts of change through time and across spatial scales (Rocheleau 1995,Jiang 2003,Nightingale 2003). Past research in Maasailand has led to development of relatively simple conceptual models to help analyze the critical pressure points and thresholds in changing land use and wildlife populations (Sinclair & Arcese 1995,Homewood *et al.* 2001,Serneels & Lambin 2001b). These studies have however, pointed the need to further understand what are the main determinants shaping livelihoods and triggering change. To what extent do external factors such as biophysical and eco-climatic/agro-ecological factors on the one hand, and infrastructure and policy on the other, shape livelihood choices? To what extent are livelihoods determined by socio-demographic characteristics of the household? What trends do these patterns indicate in terms of land use change, poverty trajectories and wildlife conservation?

Several studies globally investigated drivers of land-use change (Serneels 2001,Lambin *et al.* 2001,Burgi & Russell 2001,Geist & Lambin 2002). However, only a few have tried to link socio-political historical changes to biophysical impacts of land-use change (Reid *et al.* 2000,Stokes *et al.* 2008). A major reason for researching historical land use change is that by understanding the past we can better understand and anticipate future trajectories (Lambin and Geist, 2006). The most significant historical change in land-cover has been the expansion of agricultural lands. The past century witnessed over half of the increase in agricultural lands worldwide, and in the developing world, half of the land-cover conversions occurred in just the past 50 years (Lambin and Geist, 2006). Research on the causes of land-cover change from global to regional levels indicated that the main drivers of change at the global level are human population, level of affluence and level of technology, while the primary drivers at the regional level are rural-to-urban migration, economic growth, changes in lifestyle, and changing economic and political arrangements (Reid *et al.* 2000,Lambin *et al.* 2001). Other causes of change include the role of institutions and influence of local culture (McCusker 2004).

Over the past four decades there has been a notable change in land-uses in the Maasai-Steppe of Northern Tanzania, especially from small-scale subsistence cultivation to extensive large-scale farming (Borner 1985,Mwalyosi 1992b,OIKOS 2002). This has resulted into a growing concern about the sustainability of the Maasai Steppe as an ecological system able to support large populations of diverse and often migratory species of wildlife and livestock (Ecosystems Ltd. 1980,Mwalyosi 1991b,Kahurananga & Silkilwasha 1997,TWCM 2000). Some of these changes have been influenced by political factors but the linkages between policies and ecological changes are still poorly understood. Notable land conversions to agriculture by pastoralists in Maasai-Steppe are linked partially to issues of land tenure, insecurity, and livelihood needs, particularly, the need of the poor and the most vulnerable families (TNRF 2005,Sachedina 2008). These and other factors negatively affect the population of large migratory wildlife as well as the livelihoods of the local Maasai communities who are almost solely dependent on their free ranging livestock both economically and culturally. Many authors have reported that declining mobility of pastoralists leads to environmental degradation and increased poverty (Campbell 1999,Talle 1999,BurnSilver *et al.* 2008). Overgrazing and land degradation occur to a greater extent when livestock is forced to stay in a restricted area thus exerting persistent heavy grazing pressure, reducing the root stock available and accelerating soil erosion (Boone 2005). Conversely, land degradation from mobile pastoralism is often temporary, allowing sufficient time for resilient vegetation to regenerate during seasons without grazing (Groom 2007).

Flexibility and mobility of pastoral livestock are essential to the sustainable utilization of the pastoral rangelands of Tarangire ecosystem but are getting increasingly constrained by the expansion of large-scale commercial and extensive but small-scale cultivation and pastoral settlements necessitated by the expanding human population. Sedentarization of the formerly nomadic pastoralists into villages has been associated with intensification of land use, deterioration, fragmentation and loss of key dry-season grazing areas and watering points (Igoe 2000,Kibebe 2005).

In this study, an ecological and socio-political approach is adopted to analyze the drivers and impacts of land-use change on the Tarangire ecosystem located in the Maasai-Steppe of Northern Tanzania due to its importance to large migratory wild herbivores and the local pastoral economy. The integrated approach will further enhance understanding of both the root causes and the underlying driving forces of land-use change in the Maasai-Steppe. More fundamental driving forces such as policies and land tenure are indirectly reflected. The approach used here will facilitate analyses of the implications of government policies and adjudication on land use which requires a detailed analysis of the policies, their timing, their actual implications and geographic impacts (Serneels & Lambin 2001c). This is one of the first studies to use an integrated approach in analyzing the drivers of land-use change and their impacts in extensive pastoral lands. The approach is applied to address the following objectives: (1) establish spatial and temporal patterns of land-use changes due to agriculture using remote sensing data derived from satellite imagery, (2) analyze agricultural expansion in relation to rainfall zones and proximity to protected areas and (3) review the major historical changes and time lines related to land tenure and land use as influenced by governmental policies, (4) evaluate impacts of expanding agriculture on migration routes and habitats for wildlife and livestock in the Tarangire ecosystem.

2.2 Materials and Methods

2.2.1 Study area

The study area (Fig. 2.1) is the Tarangire ecosystem (Lamprey 1964), which is defined by the movements of its migratory animals, and consists of Tarangire National Park (TNP) and its dispersal areas in Monduli and Simanjiro districts (Borner, 1985, Prins, 1987). The area rises from about 1,000 m (asl) in the southwest to 2,660 m (asl) in the northeast. It has a bimodal rainfall averaging 650 mm per annum, with short rains from October-December and the long rains from March-May. The rains, particularly the short rains, are very unreliable and often fail.

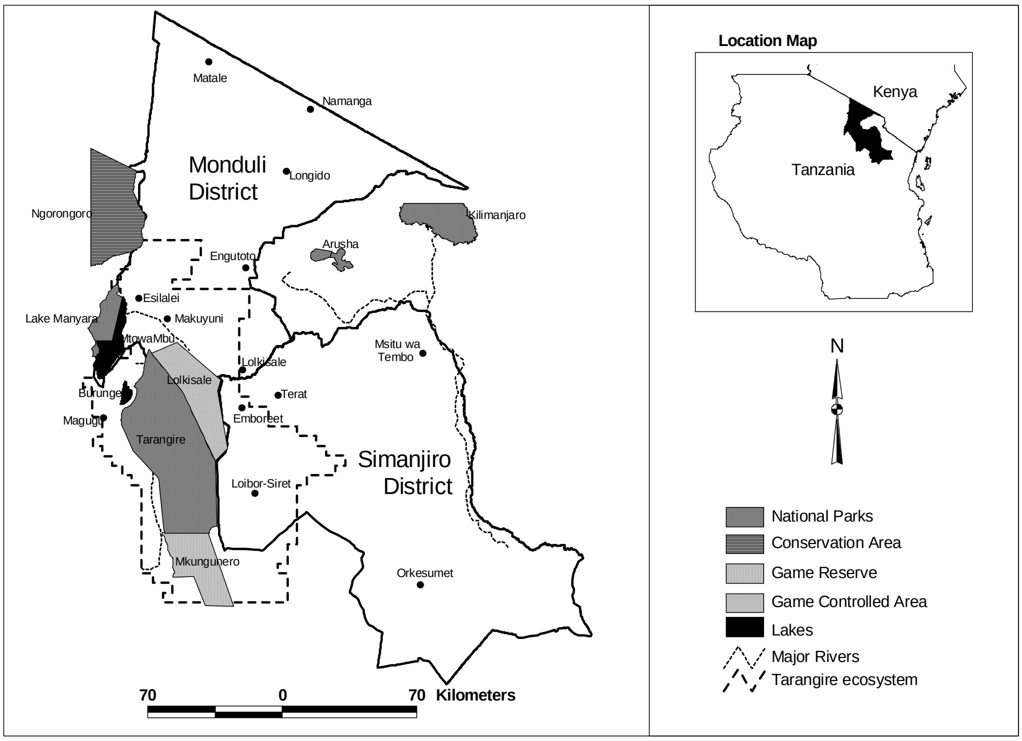


Figure 2.1 Map of the study area with the administrative boundaries of Monduli and Simanjiro Districts, core Tarangire National Park, neighboring protected areas and location in northern Tanzania.

The Tarangire ecosystem hosts the second-largest population of migratory ungulates in East Africa and the largest population of elephants in northern Tanzania (Douglas-Hamilton 1987,Foley 2002). At the onset of the rains the large mammals disperse to areas outside the park (Lamprey 1964) to the Simanjiro plains, an important wet season dispersal and calving range for wildebeest (*Connochaetes taurinus*) and zebra (*Equus burchelli*) (Kahurananga and Silkilwasha, 1997; see Plate 2.1). Factors driving these migrations are not fully understood but animals are probably attracted to these areas (Kahurananga & Silkilwasha 1997) because they are richer in minerals (TCP, 1998) and forage to satisfy the high energy demands of lactation (McNaughton 1990).

The system is also heavily utilized by Maasai livestock including cattle, sheep, goats and donkeys (see Plate 2.2) of which an estimated one million zebu cattle (Patel, Unpub. Report) constitute about 90 per cent of the grazing animal biomass (500 kg/km^2) on the Simanjiro Plains in the dry season (Mwalyosi 1992b). Like many of the unprotected lands in Tanzania, the Tarangire ecosystem, particularly the Simanjiro Plains, are under pressure from expanding cultivation, which increasingly excludes wildlife and livestock (Voeten 1999).

a)

b)

Plate 2_1. a) Key migratory species (wildebeest and zebra) in Tarangire-Simanjiro ecosystem. Note expanding cultivation along their migratory routes in b) towards the park on the background with noticeable thick and intact vegetation

a)

b)

Plate 2_2. a) Livestock (cattle) herding during the dry season in Simanjiro plains, note the bare ground and degraded environment b) one of the remaining watering points in Emboreet (Simanjiro plains) during the drought of 2005/06

2.2.2 Methods

Analysis of the drivers of land-cover and land use change requires use of multiple methods and critical interpretation of the data to characterize the drivers and impacts of change through a hierarchy of temporal and spatial scales (Reid *et al.* 2000,Campbell *et al.* 2005). An ecological and socio-political framework was adopted to analyze and evaluate the underlying drivers, causes and impacts of land-use change in the Tarangire ecosystem using historical and contemporary data obtained from archives, remote sensing, interviews, meetings, field work and observations as well as from other research work as outlined below.

(i) Classification of satellite imagery

Land cover change was analyzed using two Landsat TM scenes (Path/Row 168/62 and 168/63 of December, 1984) and two corresponding Landsat ETM+ scenes (February, 2000) acquired from the United States Geological Surveys (USGS). The following considerations dictated the choice of the images. First, a search for images matching the timing of the major policy changes and/or events related to land-use changes in the study area over the preceding 15-20 years. Second, the images should reflect similar vegetation conditions. The 1984 images were acquired during the short rainy season in December, when rainfall averaged 133 mm. The 2000 images were taken during the short dry season in February, when rainfall averaged 62 mm, well after the short rainy season (Oct-Dec.) The Normalized Difference Vegetation Index (NDVI) data from the study area suggest that both images depict comparable conditions since the NDVI for both was just above zero (Los (1998); Fig. 2A.1)).

The study area was clipped from the mosaic and cloud masked pairs of imagery in Erdas Imagine version 8.7 and next performed two unsupervised classifications, with 100 classes for each of the years (Fig. 2.2). The unsupervised 2000 image was then subjected to supervised classification while using on the 2000 Africover land-cover map. No land-cover maps were available for 1984 or around this period. Hence, the 2000 image characteristic was used to classify the 1984 image for consistency. Areas of similar characteristics in both images were visually identified, and for these areas the class of the 2000 image was assigned to the 1984 image. Finally the classified images for both 2000 and 1984 were reclassified into 8 and 10 broad land-cover classes, respectively (Fig. 2.3; also see Appendix 2B for details).

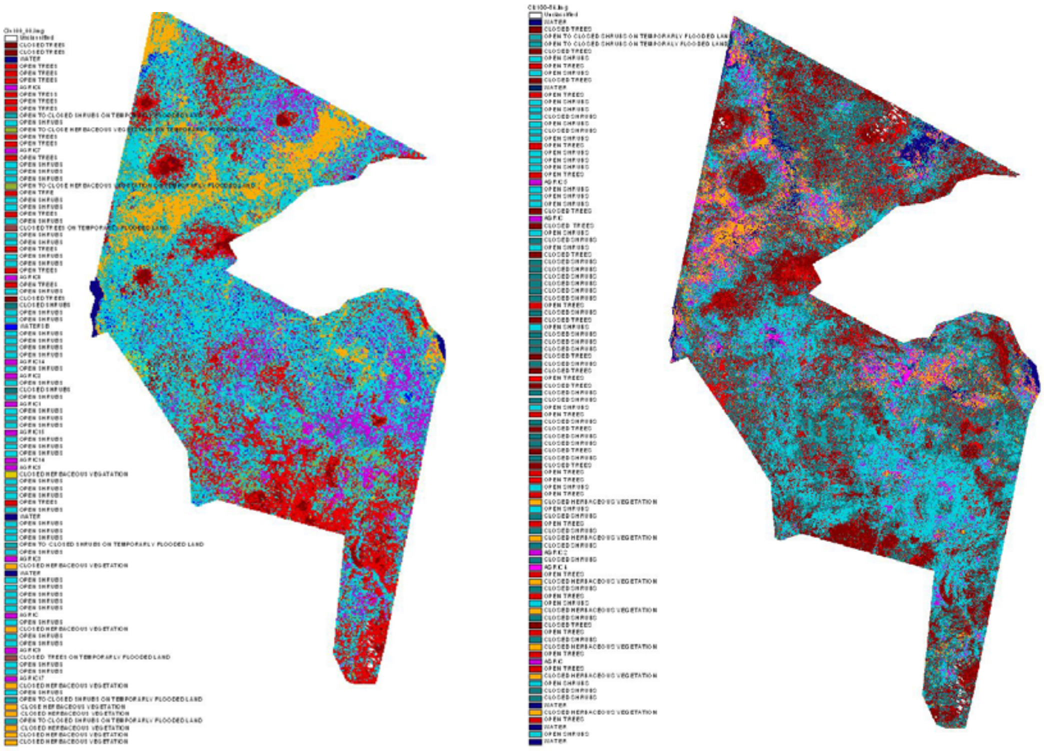
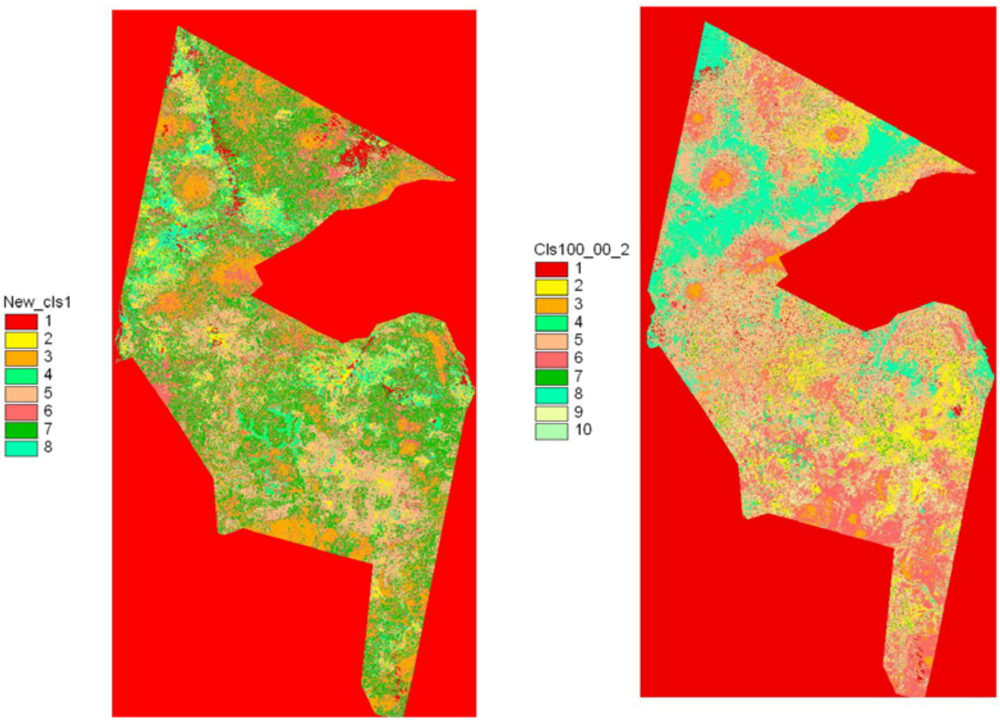


Plate 2_3 Images showing the complete classification of the 2 images into 100 classes

These images were later classified into 8 (1984) and 10 (2000) broad land-cover classes as shown below:



1984 broad land-cover classes 2000 broad land-cover classes

Plate 2_4 Images showing the merged classified broad land cover classes for 1984 and 2000

1984 Code	2000 Code
1- Water 2- Agriculture 3- Closed Trees 4- Open to Closed shrubs on temporarily flooded land 5- Open Shrubs 6- Open Tress 7- Closed Shrubs 8- Closed herbaceous vegetation	1- Water 2- Agriculture 3- Closed Trees 4- Open to Closed shrubs on temporarily flooded land 5- Open Shrubs 6- Open Trees 7- Closed Shrubs 8- Closed herbaceous vegetation 9- Open to closed herbaceous vegetation on temporally flooded land 10- Closed trees on temporally flooded land

To map agricultural land use change between 1984 and 2000, the following three classes for land use were identified: (1) areas with agriculture in 2000 but not in 1984, (2) areas with agriculture in both 1984 and 2000 and (3) areas with agriculture in 1984 but not in 2000. Fieldwork was conducted to validate the results of this classification. With the help of Maasai elders knowledgeable about historical land use, the presence of agriculture in 1984 and 2000 was ascertained and accuracy assessment of the images classification was performed according to (Congalton & Green 1999).

(ii) Drivers of agricultural expansion

The Almanac Characterization Tool (ACTS) database (Mud Springs Geographers 2002) was used to extract spatial rainfall data to define three broad rainfall zones: < 500 mm (zone 1), 500-600 mm (zone 2) and > 600 mm (zone 3). Differences in the rate of agricultural expansion between 1984 and 2000 were performed by rainfall zones using the Mann-Whitney-Wilcoxon signed rank test in SPSS (SPSS, 2005 Version 1).

Long-term monthly rainfall data (1960 to 2007), collected from one station for each district, obtained from the Tanzanian meteorological department, were analyzed for trends and seasonal and annual variation components using standardized anomalies ($z = (x_t - \bar{x})/\sigma$), where x_t is the rainfall component in year t, $\bar{x}$ is the mean and σ is the standard deviation of the rainfall. Rainfall totals falling within the percentiles 0-10, 11-25, 26-40, 41-75, 76-90, 91-95 and 96-100% were classified, respectively, as extreme, severe or moderate, drought years, normal, wet, very wet or extremely wet years (Ogutu *et al.* 2008).

Distribution in agriculture was analyzed as a function of distance from protected area boundaries using buffers of 500 m in ArcView GIS v 3.2 to test the hypothesis whether the rate of expansion in agriculture increased with distance away from parks.

(iii) Impacts of agricultural expansion

The impacts of agriculture on the wet season range and migratory routes of wildlife was analyzed by overlaying (in ArcView GIS v 3.2) maps of the distribution of 1984 and 2000 agriculture with maps of historic (Lamprey 1964a) and recent wildlife corridors/routes, livestock grazing areas and wet season ranges for the key migratory wildlife species (based on distribution maps derived from various aerial and ground censuses).

(iv) Ecological time-lines

Ecological time-lines (Stokes *et al.* 2008) were used to summarize major socio-ecological and political events that occurred in Tarangire ecosystem and the larger Maasai-Steppe, from the late 19th century to the present. The timeline was broken down into five major periods: the pre-colonial (before 1880); the colonial (1880-1950), the independence (1960-1970), the post-independence (1970-1990) and the contemporary (2000-2009) periods. Information to develop the time-lines was collated from various sources, including literature review, archives, field observations and informal interviews with long term residents and experts who have worked in the area. Ideas and views collected from these sources were presented in a workshop; the comments, suggestions and insights from participants refined the identification, timing and impacts of major changes and events underlying land use change.

2.3 Results

2.3.1 Broad trends in land-use change from remote sensing: agricultural expansion

The land under agriculture increased five-folds from 170 km^2 (17,000 ha) in 1984 to about 881 km^2 (88,100 ha) in 2000 (Table 2.1). Almost 3% of the total land area was under agriculture in 2000, up from less than 0.6% in 1984. However, agricultural expansion from 1984 to 2000 was not spatially uniform and increased 7.5 times in Monduli district compared to 3.5 times in Simanjiro district over this period (Fig. 2.2.a, b).

The expansion in agriculture between 1984 and 2000 was statistically significant for the entire study area (Wilcoxon rank test, Z = 142.4, P <0.0001) and its constituent districts of Monduli (Wilcoxon rank test, Z = 93.7, P<0.0001) and Simanjiro (Wilcoxon rank test, Z = 93.6, P<0.0001). Also, mean field size was significantly larger (Wilcoxon rank test, Z = 32.2, P <0.0001) in Simanjiro (ca. 4.2 ha) than Monduli (ca. 1.4 ha) in both years. Also, mean field size over the entire study area was about 1.05 ha in 1984 but increased to 1.96 ha in 2000. Land-use changes indicated that about 75% of the area under agriculture in 1984 was abandoned and was not under cultivation by 2000 (Fig. 2.2.c, d). Most of the abandonment occurred in Simanjiro (79 %), the drier district, compared to Monduli (48 %), the wetter district that also experienced a higher increase in the number of people (Table 2. 3).

Table 2.1 Changes in cultivated area (in km^2) between 1984 and 2000 in the Tarangire Ecosystem, Monduli and Simanjiro Districts. The changes show areas where agriculture was present in 1984 and 2000, abandoned by 2000 and where new fields were created by 2000.

	Agriculture area (km^2) in 1984	Agriculture area (km^2) in 2000	Unchanged area (km^2)	Abandoned area (km^2)	New fields area (km^2)
Study Area	170	881	42 (25%)	128 (75%)	739 (84%)
Monduli	70	520	21 (30%)	49 (70%)	499 (96%)
Simanjiro	100	362	21 (21%)	79 (79%)	341 (94%)

2.3.2 Agricultural expansion in relation to rainfall zones and rainfall patterns

At the district level the relation between agriculture expansion and rainfall was spatially variable (Fig. 2.3). The expansion partly follows the rainfall zones but also mirrors the long-term rainfall trend (Fig. 2.4). Agriculture increased significantly in all rainfall bands (Table 2.2) but the rate of increase was larger the higher was the rainfall band and was 3, 15 and 30% in the lowest (<500 mm), medium (500-600 mm) and highest (>600 mm) rainfall bands, respectively. In Monduli high increases in agriculture occurred in rainfall bands >600 mm (9%) and 500-600 mm (4.6%) and moderate increases in <500 mm (2%) rainfall band, while in Simanjiro high increases occurred in lower bands of less than 500 mm and moderate increases in the highest rainfall band (>600 mm). Hence agricultural expansion in the study area is not solely determined by rainfall distribution. Other drivers apparently play influential roles as well.

One of the key drivers of land use change is climate. The long-term rainfall in Tarangire-Simanjiro ecosystem showed a 3-5 year quasi-periodicity (Fig. 2.4), meaning that there is either a drought or below-average rainfall condition every 3-5 years and implying frequent crop failures. Rainfall was highest during 1962-1970, while the 1971-1977 period showed declining rainfall, with a severe decline evident in rainfall between 1974 and 1976. After 1976, annual rainfall increased up to 1979. Between 1979 and 2007 the frequency of dry and wet conditions increased significantly compared to the earlier decade. During the study period extreme droughts occurred in 1961, 1965, 1974, 1976 and 1991 and severe droughts in 1967, 1975, 1982, 1992, 1993, 1997 and 2003. Moderate droughts were registered in 1963, 1971, 1972, 1977, 1983, 1994 and 2004. Extremely wet years occurred in the study area in 1968 and 1998, while very wet years in 1962 and 1979. Wet years were 1964, 1966, 1978, 1989, 1990, 2000 and 2001. The extensive cultivation in parallel with farm abandonment observed in the Tarangire-Simanjiro area is therefore not surprising; given the increasing climatic variability over the past two-three decades.

Table 2.2 Results of Mann-Whitney U-tests comparing changes in areas under cultivation by district and rainfall zones. Note the significant relationships between increasing cultivation and rainfall.

District	Rainfall band	1984 vs. 2000	
Monduli	<500	14.2	(P<0.0001[a])
	500-600	6.5	(P<0.0001)
	>600	110.5	(P<0.0001)
Simanjiro	<500[b]		
	500-600	50.5	(P<0.0001)
	>600	113.6	(P<0.0001)

[a]Probabilities are two-tailed and based on the normal approximation. [b]Absence of agriculture in the portion of Simanjiro in the <500 m rainfall band mapped in 1984 precluded pair wise comparisons.

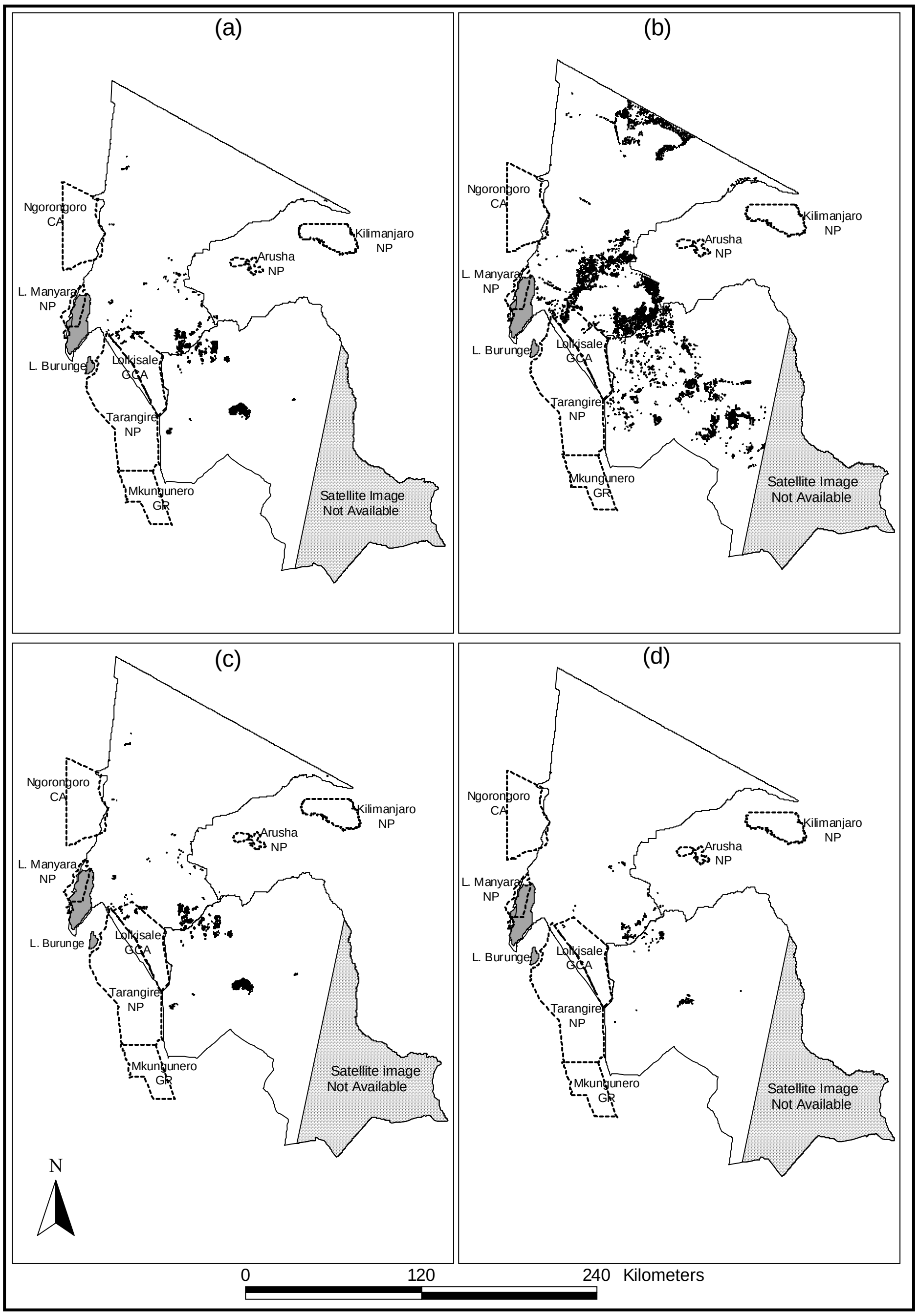


Figure 2.2 Land-cover changes in the Monduli and Simanjiro districts of the Tarangire Ecosystem showing (a) agriculture in 1984, (b) agriculture in 2000, (c) areas where agriculture was present in 1984 but not in 2000 and (d) areas where agriculture was present in both 1984 and 2000.

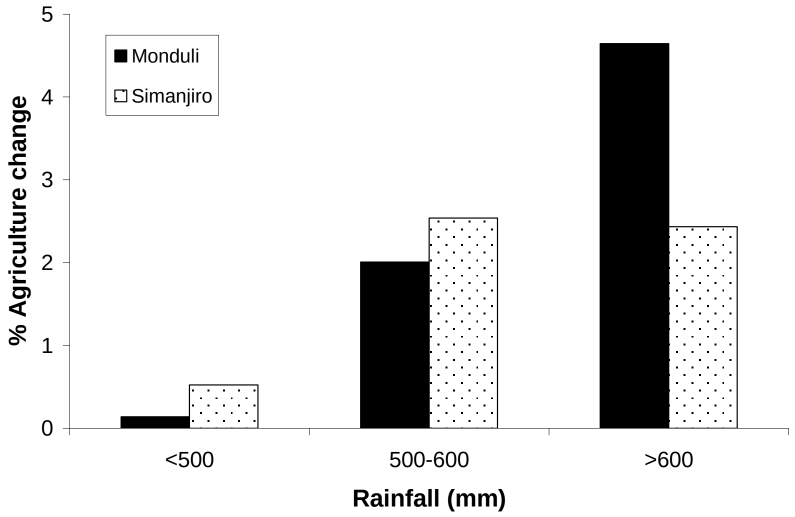


Figure 2.3 Percentage changes (annual) in cultivated area in relation to low (<500 mm), moderate (500-600 mm) and high (>600 mm) rainfall zones in Monduli and Simanjiro Districts between 1984 and 2000.

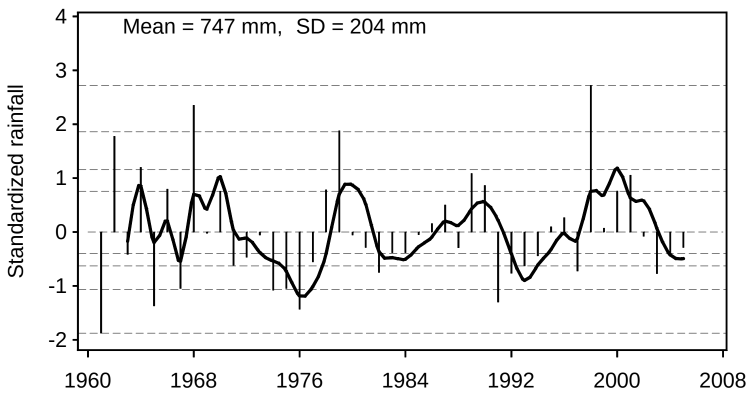


Figure 2.4 Annual rainfall pattern in Tarangire-Simanjiro. Vertical lines (needles) indicate the standardized values and solid lines are the 3-5 year running means. Dashed horizontal lines are 10, 25, 40, 75, 90, 95 and 100th percentiles

2.3.3 Agricultural expansion in relation to protected areas and its impacts on wildlife migratory corridors

Tarangire ecosystem boundary defines the spatial extent covered by wildlife migrating from TNP to dispersal and calving areas during the wet season (Fig. 2.1). At about 40 km it is the distance where majority of the wildebeest and zebra concentrate for calving, while 60 km is the maximum distance that majority of the key migratory species can reach. In the 1980s agriculture was limited both in its distribution and in the size of farms. Although fields were located less than 40 km from protected area boundaries in the ecosystem at times, many of them were smaller in size, averaging 3.4 km^2 (Fig. 2.5.a) compared to 2000 when majority of the fields located less than 40 km from protected area boundaries averaged more than 15 km^2 in size (Fig. 2.5.c). This is almost a five-fold increase in farm size between 1984 and 2000. Cultivation expanded, both away from and towards protected areas, from about 4 % in 1984 (Fig. 2.5.b) to more than 7 % by 2000 (Fig. 2.5.d), corresponding to an increase in the absolute area under cultivation of approximately 725 km^2 between the two periods. Fewer and smaller fields were observed inside Lolkisale GCA in 1984 (Fig. 2.2.a), with a few scattered north of TNP and less than 20 km from the park boundary. This changed drastically in 2000 both in the size and location of farms, with increased cultivation inside Lolkisale GCA and towards TNP (Fig. 2.2.b & 2.5.c).

Extensive cultivation has occurred on the eastern sections of TNP with intensive concentration of farms in the northeast, between Emboreet and Lolkisale (the calving area) towards Makuyuni and LMNP in Esilalei, Mto wa Mbu and Engutoto. This is reflected in the increase of fields in 2000 beyond 60 km as indicated in Fig. 2.5.c relative to 1984. Hence contrary to the hypothesis cultivation has expanded both ways, towards and away from protected areas boundary.

Cultivation has taken up the traditional migratory routes, particularly on the north and north-west of TNP. Fig. 2.6 shows the spatial extent of agricultural expansion from the 1980s (Fig. 2.6.a) to 2000s (Fig. 2.6.b). On the eastern side of TNP, the routes towards Lolkisale GCA and Simanjiro plains are threatened with blockage by expanding cultivation. Furthermore, the wet season range for the key migratory species continues to shrink as more of the rangeland is converted to farmland.

Accordingly, the densities of wildlife species mapped from aerial surveys (wet season) for the two periods revealed a downward trend (Fig. 2.6). Likewise for the livestock, their grazing land is being taken up by cultivation.

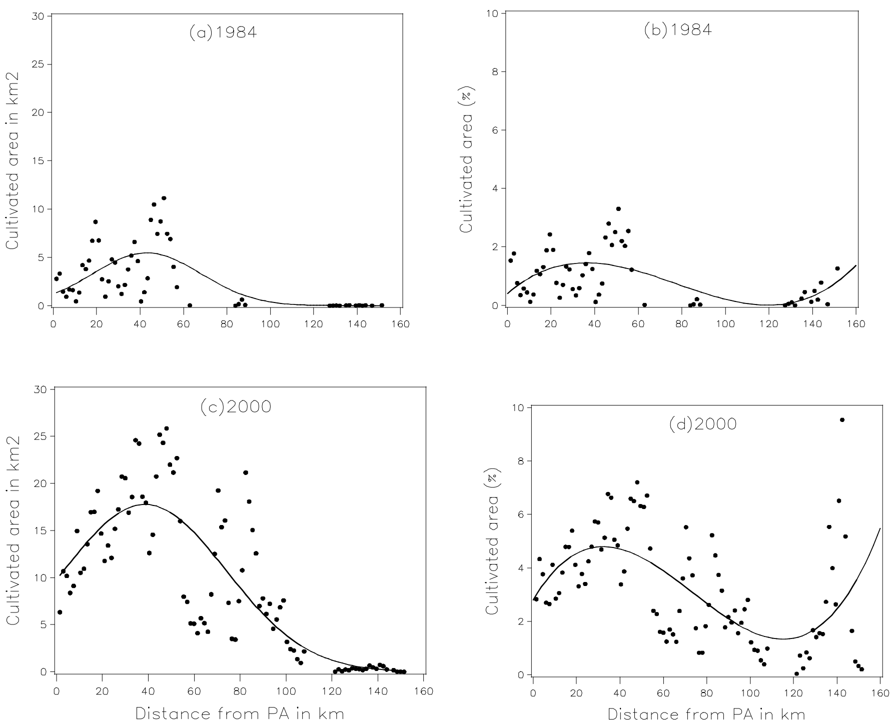


Figure 2.5 Cultivated area as a function of distance from protected areas boundary in 1984 and 2000; (a & c) absolute cultivated area (in km^2) from the nearest protected area boundary and (b & d) cultivated area as a percent of available area from the nearest protected area boundary

2.3.4 Historical analysis and synthesis of policies and major events that have impacted land use changes in Tarangire-Simanjiro ecosystem

Current land-use changes observed in the Tarangire-Simanjiro ecosystem emanate from a number of historical events which have disrupted the traditional cultural and social practices of the Maasai pastoralists. Below is a review of some of the major historical events in the Maasailand over the last century which are closely associated with the drive for land-use change observed in the study area. For clarity, these major events are organized into the following five periods;

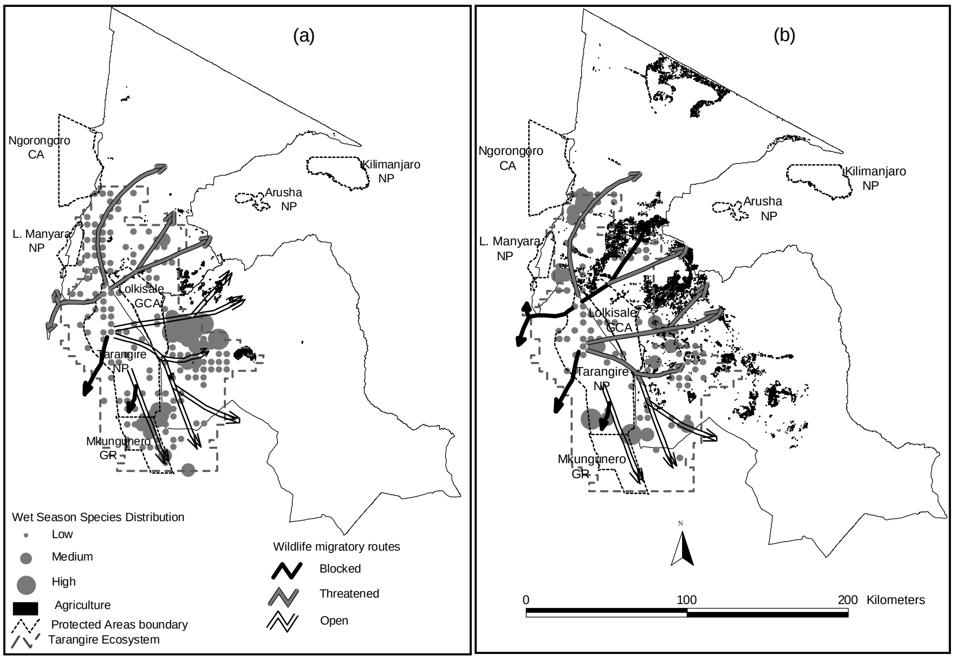


Figure 2.6 Wet season distribution of migratory wildlife species; (a) during 1984-1994 in relation to agriculture in 1984 and (b) during 1997-2004 in relation to agriculture in 2000.

(i) The pre-colonial period: before 1880s.

Between the 16th and 18th centuries the Maa-speaking people expanded their influence from Lake Turkana in northern Kenya, southwards throughout the Rift Valley area to modern Tanzania Maasailand replacing other pastoral groups like the Nilotes and Bantu who were also cultivators (Homewood & Rogers 1991). By 1880 the Maasai reached their greatest extent both in terms of numbers and influence. It was not before the turn of the 18th century when they were hit by pleuro-pneumonia and small pox diseases that killed many of them. At the same time the outbreak of the rinderpest in cattle and wildlife decimated Maasai livestock at around the beginning of the 19th century (Coast 2002). This meant that the main livelihood, i.e. livestock was gone. With no options for replenishing their depleted stocks, many of them started crop cultivation.

(ii) The colonial period: 1880s – 1950s.

The 19th century had another big influence in terms of changing pastoral way of life as was the colonial period that saw the displacement of Maasai from high potential land for agricultural development by the European farmers/settlers (Homewood & Rodgers, 1991). This move started in northern Kenya by the British colonialists and moved gradually southerwards to German Tanganyika in Tanzania. Many of the game parks were created at the same time through the eviction of pastoralists from key resources such as the dry season grazing areas and watering points. Because of the abundant water and pasture in the Tarangire-Simanjiro ecosystem, it had a reputation as one of the best pastoral areas in Tanzania. Many herders who were evicted from the Serengeti National Park in the 1950s relocated to this area (Igoe 2000).

(iii) Independent period: 1960s-1970s.

Tanzania became independent in 1961 and most of the British colonial administration/legislation was adopted by the new government. In this period rinderpest that had previously killed many wildlife and livestock species was controlled. The control of rinderpest boosted the numbers of wildebeest in the Serengeti ecosystem. This further pushed Maasai to the south, with some moving into Tarangire-Simanjiro area to avoid contact with wildebeest calving areas in the short grass-plains. Such areas are associated with the spread of Malignant Catarrhal Fever (MCF) that affects cattle. Between 1962 and 1963 the worst drought in 50 years hit most parts of the country including Tarangire-Simanjiro area and killed many wildlife and livestock (Lamprey 1963). In 1967 the first president of Tanzania, the late Dr Julius Nyerere, declared Tanzania a socialist country, through the Arusha Declaration that put a nationalization policy in place (Kikula 1997). Many of the settlers' farms/plantations were nationalized by the government and managed by parastatals like the National Farming Company (NAFCO) and National Ranching Company (NARCO). Agriculture was promoted as the back-bone of the national economy. Large-scale farms like the Lolkisale seed-bean farms were established in Tarangire-Simanjiro to produce crops for export as well as for national reserves during droughts and food shortage. Data from the Tanzania National Bureau of Statistics (NBS) indicated a high population increase in the study area in the past 20+ years (Table 2.3). The growth was due both to natural increase and immigration from nearby regions of Kilimanjaro and Arusha whose inhabitants were mainly agriculturalists (Igoe 2000). This move continued to displace Maasai pastoralists from their best rangelands into more marginal areas.

(iv)The post independent: 1970 – 1990s.

In 1970, the Tarangire Game Reserve was upgraded to become Tarangire National Park including the southern portion of approximately one third of the area in the northern Mkungunero. People who were residing in this area had to move to the Simanjiro plains as there were still sufficient pasture and watering points. In the same period the villagisation policy (1974) was introduced in the country, as a follow-up of the socialism ideology of 1967. People were forced to live in nucleated villages in order to have access to social services including schools, hospitals, veterinary, and market centers (Kikula 1997). This new policy restricted the movements of pastoralists with their livestock and forced many to settle in village land plots which were assigned primarily for cultivation, as every villager was supposed to have a plot of land for building a house and producing food for their families. Maasai herders living east of Tarangire began to feel squeezed as commercial seed bean companies and peasants from the slopes of Mount Meru and Kilimanjaro began moving into the area in the early 1980s. By the mid 1980s the movement of commercial interests and peasant farmers into the area had expanded to the villages of Central Simanjiro (Igoe 2000), blocking traditional migratory routes for wildlife (Borner 1985a). In 1990s Tanzania, as other countries in the world, was forced to embrace globalization, thus moving away from socialism to a free-market economy in which the social services formerly provided for free by the government ceased (Shivji 1998).

The economic liberalization policies including the Promotion Investment Acts of 1992 were declared. Tanzanite mining at Mererani in Simanjiro was established and attracted many young Maasai men from Simanjiro and neighboring regions (Muir 1994). This resulted in changes in lifestyles as some of the young men became very rich in a short time. Some of them invested back home by buying more livestock to replenish their herds whilst others used the money to open up new farms for cultivation using modern farm machinery instead of traditional hand-hoe. In parallel with this, many agro-pastoral people moved into Tarangire-Simanjiro as land was seen as plenty for farming (Igoe 2000). The local Maasai decided to lease out their land to these immigrants for fear of losing their land to conservation due to speculations of impending expansion of protected area boundaries like that of TNP (TNRF 2005).

(v) The current period: 2000-2009:

The National Land Act (Act no. 4 of 1999), with its follow up of the Village Land Act (Act no. 5 of 1999), gave villages more autonomy over the use of land. In the same period a new category of wildlife conservation area was agreed to village land, the Wildlife Management Areas (WMA, 2003) (MNRT 1998). Many agro-pastoralists, particularly in Tarangire-Simanjiro have been reluctant to accept the WMA concept because of past painful memories related to establishment of protected areas (i.e. eviction of people), despite the fact that they present a potential pathway for enriching their livelihood options. The new livestock policy was created in the same period following pressure to increase investments in rangelands and advocacy by local NGOs and civil societies to improve the livelihoods of pastoralists in the country.

Table 2.3 Human population changes in the Tarangire ecosystem, Monduli and Simanjiro Districts between 1978 and 2002. The figures show annual percentage population growth rates between 1978-1988 and 1988-2002.

	1978	1988	2002	Annual growth rate (%) 1978-1988	Annual growth rate (%) 1988-2002
Study Area	145441	201357	325652	3.3	3.4
Monduli	121784	148460	184516	2.0	1.6
Simanjiro	23657	52897	141136	8.0	7.0

Source: Tanzania National Bureau of Statistics, 2007.

Figure 2.7 Time-lines of the major policies and historical events in Tanzania which have shaped land-use change in the Tarangire-Simanjiro Ecosystem (adapted from Stokes, *et al.* 2008)

2.4 Discussion and conclusions

This study has shown the rapid conversion of rangeland to agriculture in the dispersal areas of Tarangire National park. The remote sensing data revealed that the area under agriculture increased from 170 to 881 km^2, equivalent to $10\%.yr^{-1}$. This increase in cultivated land was due to large-scale farming, such as the cultivation of seed beans in Lolkisale, and small-scale subsistence farming.

With an overall increase of the area under agriculture, one might expect this to happen particularly in areas further away from the park boundary, as they are typically having fewer infrastructures and more remote from markets. The remote sensing and ground observations revealed the contrary: agriculture evidently expanded both away from and closer to the protected area boundaries.

The contribution of population change as a driver of land use change was also investigated. The human population growth rate of the whole area of $3.4\ \%.yr^{-1}$ was much lower than the annual increase in agricultural area of $10.3\ \%.yr^{-1}$. More strikingly, Simanjiro, the district with highest population growth rate ($7.4\ \%.yr^{-1}$) had a lower rate of agricultural lands (8.0 %.yr) than Monduli ($1.7\%.yr^{-1}$ and $12.5\ \%.yr^{-1}$ respectively). There is no doubt that population increase, driven by natural growth and immigration, have been an important driver of the observed land use change. The analysis reveals, however, that human population growth alone does not explain all.

Analysis of long term trends of rainfall revealed no evidence to support holding climate change responsible for the observed land use change. There was a cyclic pattern of 3 to 5 years in annual rainfall totals, but not a marked trend over longer time periods. Hence, the data does not provide the evidence to support the hypothesis that climate has been a driver for long term land use change. Droughts, however, are likely to exert an influence on short term land use dynamics. The land use change described here was not permanent; a large portion of the land under agriculture in 1980 was abandoned in 2000. This might be attributable to the 3-5 year quasi-periodic oscillation in rainfall, because more than 90% of the farming is rainfall dependent (Kibebe 2005). The recurrent droughts preclude permanent cultivation as a sustainable livelihood option, because crop failure is frequent. Agro-pastoralism, which combines cropping and livestock production and allows falling back on livestock resources in case of drought, might be a more viable livelihood option. It is thus advisable to minimize the land converted to cultivation and instead diversify livelihood options by establishing community conservancies (WMA) to enhance revenue flows from wildlife and wildlife-based tourism both of which are more compatible with pastoral livestock rearing and more resistant to frequent droughts.

Furthermore the study has shown that the lands converted to agriculture increasingly block the migration of wildlife. Consequently, the number of routes between TNP and its dispersal areas that have remained open to migratory wildlife has declined from 9 to 5 over the past 25 years, and even these 5 remaining routes are threatened to be blocked by the expanding cultivation and settlements around the park (Borner 1985a,OIKOS 2002b,Gereta *et al.* 2004). Also, the traditional wildebeest calving grounds on the Simanjiro Plains have been taken up by the expanding cultivation.

Above, this study has shown that there is little evidence for climate change. However, the shrinking wet season range might amplify, similarly to what has been suggested for elsewhere (Serneels & Lambin 2001b,Ogutu *et al.* 2008) the influence of droughts on wildlife and livestock, leading to marked population declines and delayed recovery from droughts due to food insufficiency. It has been suggested that the land use change adversely impacted the livelihoods of the agro-pastoralists in Tarangire ecosystem, due to reduced livestock per capita related to the loss of pastoral grazing land (Sachedina 2008). It has been suggested that other factors such as policy, land tenure and land potential affect land-use change as well (Homewood *et al.* 2001,Campbell *et al.* 2005). The biophysical suitability of land is a strong determinant of the conversion of bush to agriculture. Changes in land-tenure, driven by governmental policies since the colonial era, have played a significant role in land-use change across the Maasai-Steppe and particularly in the Tarangire ecosystem. In the pre-colonial period, when the land was communally owned and resources abundant, pastoralists were few and ranged freely, which allowed sustainable use of rangeland resources and co-existence with wildlife (Peterson 1978a,Voeten 1999). During the colonial period large-scale plantations, excluded pastoralists to some of their previous key grazing lands (Igoe 2000). After independence policies continued encouraging agriculture at the expense of pastoralists (Shivji 1998). The Villagization Policy of 1974, which forced people to live in nucleated villages, further enhanced sedentarization of nomadic pastoralists. The Investments Act of 1992, which encouraged investment in mining in rangelands, further marginalized the people of

Tarangire ecosystem. The Land-Acts of 1999 that gave villagers more autonomy over land use (Tenga *et al.* 2008) further accelerated sedentarization of the pastoral communities. These changes in land-tenure led to rapid population growth, as a result of autochthonous growth and immigration of people from other regions seeking arable land (Mwalyosi 1991c,Campbell 1999) and young people attracted by mining (Igoe, 2000), Fig. 2.8.

These processes have resulted in the reported land use change. The blockage or loss of the traditional migratory routes increasingly undermines wildlife conservation in Tarangire ecosystem. Additionally, loss of the wet season dispersal range and widespread poaching of wildlife outside the protected areas threaten the future of wildlife conservation in this ecosystem. Not surprisingly, several studies have linked the declining populations of large mammals in the Tarangire-Ecosystem (e.g. wildebeest) to the rapidly changing land use, particularly the conversion of rangelands to agriculture and overexploitation of species through hunting and poaching (Galanti *et al.* 2000,TNRF 2005). Recent aerial surveys in Tarangire-Ecosystem revealed extreme declines in numbers of the key migratory wildlife species, most notably wildebeest, whose numbers dropped from about 43,000 in 1988 to a mere 5,000 in 2001 (TAWIRI 2001). Such drastic declines undoubtedly threaten the future viability of both migratory and resident ungulate species and other species dependent on them.

This study has combined an ecological and socio-political approach in an attempt to disentangle the causes of land-cover conversions stemming from historical, political and livelihood needs in the Tarangire ecosystem. This integrated approach included both quantitative and qualitative characterizations of land-use changes and their consequences for the people, wildlife and livestock in the Tarangire ecosystem. Without such an approach it would not have been possible to comprehend the fundamental driving forces such as policies and land-tenure, which are indirect drivers of land use change. This interdisciplinary approach reveals that agriculture expanded in the past 20+ years and continues to do so.

The results of this study reinforce findings of other studies conducted in East Africa indicating that wildlife habitats inside and outside protected areas are at a high risk of becoming ecological islands able to support a fraction of the previous wildlife populations (Nelson 2007,Ogutu *et al.* 2009). The land transformations currently underway in Tarangire are similar to those observed in Kenya over the last 20 years and portend grave consequences for the future of the local migratory populations and pastoral livelihoods.

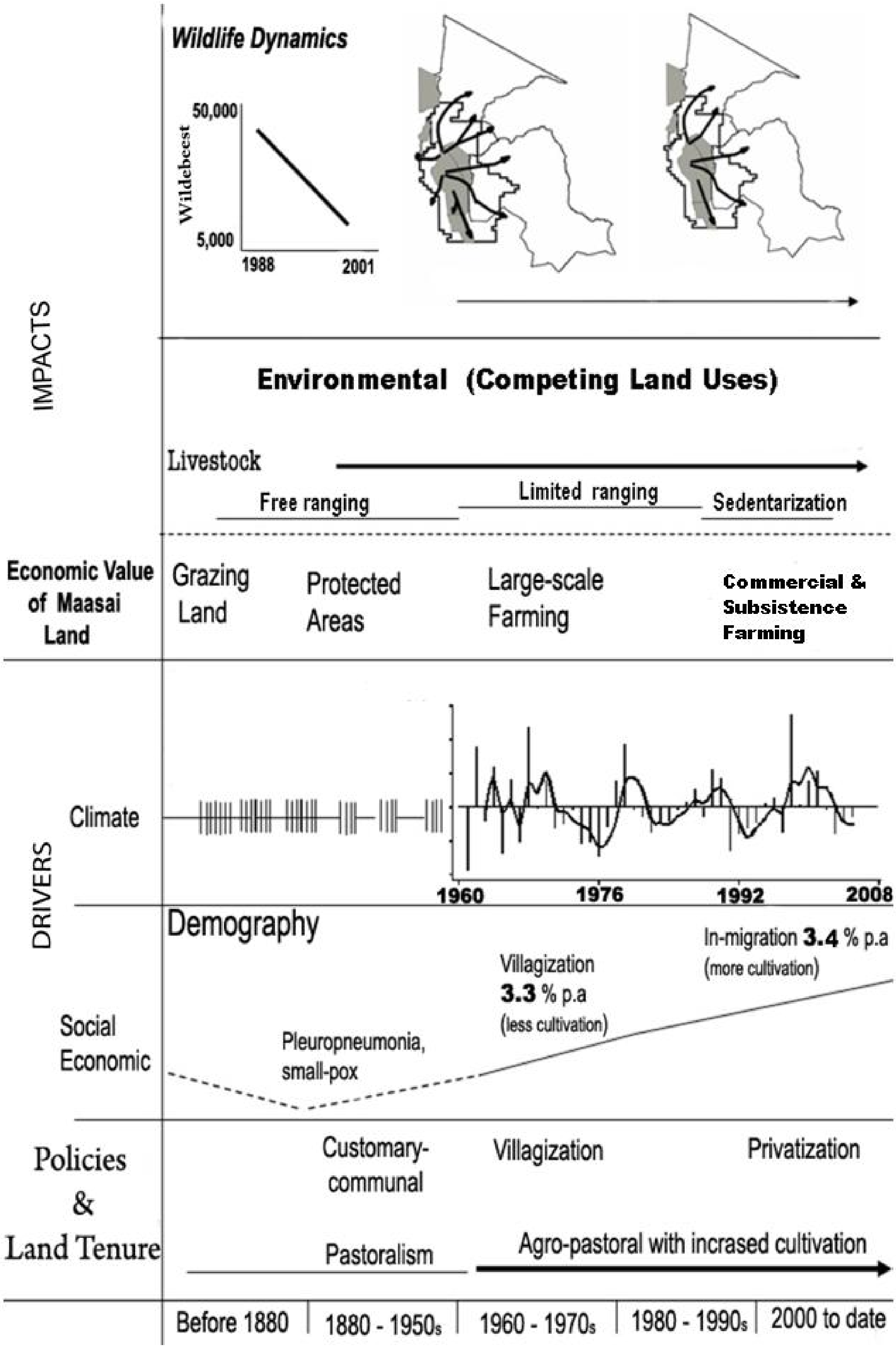


Figure 2.8 A schematic representation of drivers of change and the associated time lines, their impacts on the environment and livelihoods of agro-pastoralists in the Tarangire ecosystem (the Maasai-Steppe)

Scarcity of land and human population growth are the main drivers of the land use change, which triggered the degradation of the pastoral livelihoods and the wildlife resources. Any attempt to solve the problems of the dwindling wildlife resources, should therefore address the poverty which drives this development. Diversification of pastoral livelihoods is commonly propagated as a possible solution. However, diversification into agro-pastoralism, exposed pastoral households to crop failure risks emanating from unreliable rainfall and crop damage by wildlife. Diversification to mining improved the livelihood of a few (Sachedina 2008), but the earnings are often invested in livestock herds and expanding cultivation (TNRF 2005), which further accentuates the pressure and loss of the pastoral rangelands. In addition to the environmental risks, the vulnerability of poor pastoral households is heightened by lost access to free social services, such as education and health care (TNRF 2005), due to the change in national policy from '*ujamaa*' (socialism) to the current *free market economy* and *privatization policy*.

The current arrangement of state ownership of wildlife and governmental control of revenue streams generated from wildlife-tourism severely restricts wildlife-related options available to support the income of local people (Sachedina 2008) and diminishes the importance of wildlife conservation in local livelihood and land-use decisions (Tenga *et al.* 2008). Development of community-based tourism is one potential avenue for enhancing the importance of livestock-based wildlife conservation since community lands add a cultural value/element to tourism that is absent from the exclusively protected areas and often have just as much wildlife as do the protected parks and reserves (Kideghesho 2002,Nelson 2007). Policies that encourage development of tourism ventures on community lands would both diversify the tourism industry and encourage benefit-sharing with the local people. For this to work successfully local authorities should be empowered to manage wildlife on community lands and establish partnership ventures with private investors. Ultimately, an integrated land-use plan that considers all the different land-uses across the landscape as part of a broad development plan would be needed to minimize competition for resources and conflicts as well as sustaining these rangelands.

3 Chapter Three: Spatial correlates of land-use changes in the Maasai-Steppe of Tanzania: Implications for conservation and environmental planning[i]

Abstract

Spatially explicit models are becoming increasingly important tools for simulating land-use change. In this study, statistical models that incorporated spatial correlates of agricultural expansion were formulated and tested, and used to predict landscape-scale patterns of agricultural land-use change and its implications in the Maasai-Steppe of Northern Tanzania. Relationship between agricultural land-use and its spatial correlates was analyzed using Multiple Logistic Regression on data derived from Satellite Imageries for the year 2000. Further, examination on the implications of the agricultural land-use change on the range and migratory routes of key wildlife species in the context of wildlife conservation and land-use planning were evaluated.

Results showed biophysical variables provided the primary conditions for land-cover conversions to agriculture. There was a strong overlap between lands suitable for agriculture, wildlife migratory routes and the wet season dispersal areas. Expanding cultivation towards protected areas severely restricted wildlife movements to dispersal areas outside parks by blocking their migratory routes. Further, the global model used for the prediction of probability of land-conversions to agriculture suggested future expansions would be constrained by values of the biophysical variables analysed here. The rapidity of rangeland conversions to farming in the study area presents threat to wildlife conservation and disrupts the ecosystems viability in supporting its rich biodiversity and the agro-pastoral livelihood. There is urgency for pursuing land-use strategies and plans, which are both profitable and sustainable for the agro-pastoral communities and the wildlife. The plans should address the different land-use options by considering current and future trends, implications and the ease for their cohabitation as analysed in this study.

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[i] Materials from this chapter are currently under review for re- submission for publication as: Fortunata U. Msoffe, Mohammed Y. Said, Joseph O. Ogutu, Shem C. Kifugo, Jan de Leeuw, Paul van Gardingen and Robin S. Reid. Spatial correlates of land-use changes in the Maasai-Steppe of Tanzania: Implications for conservation and environmental planning

My contribution in this paper: I collated all the data from field, libraries, institutions and other archives. Data analyses and statistical tests were done in collaboration with Said, Ogutu and Kifugo. de Leeuw, van Gardingen and Reid advised on the analyses and commented on drafts of the manuscript.

3.1 Introduction

Environmental management and land-use planning need information about the dynamics of land use (Verburg *et al.* 2002), since land-use activities can impact significantly on natural resources. In many developing countries, particularly in the sub-saharan Africa, this information is often lacking, making planning a difficult exercise (FAO 2009). Habitat loss and fragmentation that result from land-use changes are major factors contributing to the decline of many biological populations (Dale *et al.* 1998,Salas *et al.* 2000). Forest cutting, agricultural practices (Geist & Lambin 2002,Linderman *et al.* 2005,Etter *et al.* 2006), urban and industrial expansion (Dale *et al.* 1998), road development and alteration of waterways (Houghton 1994,Li *et al.* 2004) are amongst common human land-use activities that can significantly alter the land cover (Etter *et al.* 2006) and hence adversely affect biodiversity (Serneels & Lambin 2001b,Reid *et al.* 2008,Ogutu *et al.* 2009).

Land-use change studies are central to environmental management, biodiversity conservation, ecosystem services and livelihoods (Lambin *et al.* 2000,Verburg *et al.* 2002,Turner II *et al.* 2007). The increasing availability of satellite imagery and GIS technologies allows for expanded interdisciplinary inquiry into various forces underlying land-use change and their implications (Hunter *et al.* 2003). Empirical diagnostic models of land-cover/use change can be developed from remote sensing data and used to facilitate identification of major processes of change, characterization of land use dynamics (Mertens & Lambin 1999) and anticipation of where future changes are more likely to occur. Consequently, models of land-use change combined with dynamic modelling are becoming increasingly important tools for simulating land-use change using empirically quantified relations between land-use and its driving factors (Mertens & Lambin 2000,Serneels & Lambin 2001c)

East Africa has lost more than half of its wildlife in the last 30 years (Stoner *et al.* 2006,Western *et al.* 2009b). In Tanzania, wildlife are declining in all the major wildlife areas and ecosystems, including national parks and game reserves (TNRF 2008). Most of this is driven by high human population growth in the rural areas, changing economic realities and policies (Homewood *et al.*, 2006, Norton-Griffiths and Said, *in press*, Msoffe *et al.*, *in press*). However, for wildlife to be conserved successfully outside protected areas, it should legally generate income for local communities who bear the cost of supporting wildlife. This is currently not the case in most of rural Tanzania, for example, (TNRF, 2008). As a result, many protected areas in East Africa are becoming “islands” in a sea of farms (Borner 1985,Newmark 1996).

Land use in northern Tanzania is changing rapidly and in unplanned fashion – from extensive rangelands to a patchwork containing commercial farms, subsistence plots and settlements (FAO, 2009). The growing populations, expanding economies and increasing urbanization in areas of high biodiversity demand for multiple objectives in land use planning. It also requires sufficient information to allow land managers to explore various land use options and evaluate impacts of alternative land-use strategies and the structure of trade-offs between various land uses and development objectives (Dale *et al.* 1998).

This study formulated and tested models that incorporated spatial correlates of agricultural expansion and used them to predict local- and landscape-scale patterns of distribution of agricultural land-use and its implications for wildlife conservation in the Maasai-Steppe ecosystem of Northern Tanzania in 2000. Specifically, the relationship between land-use change and its spatial correlates were evaluated using multiple logistic regression analysis (MLR) on data derived from satellite imagery for the year 2000 when agricultural expansion was high (Msoffe, et al., *In press*). MLR indicates the probability that a given grid cell undergoes land-use conversion to agriculture conditional on the set of driving factors (Mertens & Lambin 2000,Serneels & Lambin 2001c).

It is hypothesized that areas in close proximity to villages (formal settlements in rural areas of Tanzania), roads, rivers and protected area boundaries were more likely to be converted to agriculture. Further examination on the implications of the agricultural land-use change on the range and migration routes of key wildlife species were analyzed within the context of wildlife conservation and land-use planning.

3.2 Materials and Methods

3.2.1 Study area

The Maasai-Steppe is one of the richest wildlife areas in East Africa and is well known for its migration of wildebeest (*Connochaetes taurinus*), zebra (*Equus burchelli*) and elephant (*Loxodonta Africana*) (Lamprey 1964b). Tarangire National Park (TNP) is at the heart of the ecosystem and contains the Tarangire River, the only source of water in the dry season, along which majority of the large mammals, including the migratory herbivores, congregate (Lamprey 1963). At the onset of the rains these animals disperse away from TNP to areas in the north, north east and south east of the surrounding ecosystem (Lamprey 1964), Fig. 3.1.

Ecologically, the Maasai-Steppe is an important stronghold for the wildlife and pastoralists of northern Tanzania (Lamprey 1963). It contains the second-largest population of migratory wild ungulates in East Africa (second only to the Serengeti-Mara ecosystem) as well as the largest population of elephants in northern Tanzania (Douglas-Hamilton 1987,Foley 2002).

The Simanjiro plains is one of the most important wet season dispersal and calving areas for wildebeest and zebra (Kahurananga & Silkilwasha 1997,TCP 1998). Large concentrations of wildlife and domestic animals including cattle, sheep, goats and donkey share pasture in this area at various times of the year, especially the wet season (Mwalyosi 1992a,Voeten 1999). However, the rapidly growing human population, expanding cultivation and settlements in these plains are progressively excluding wildlife and livestock.

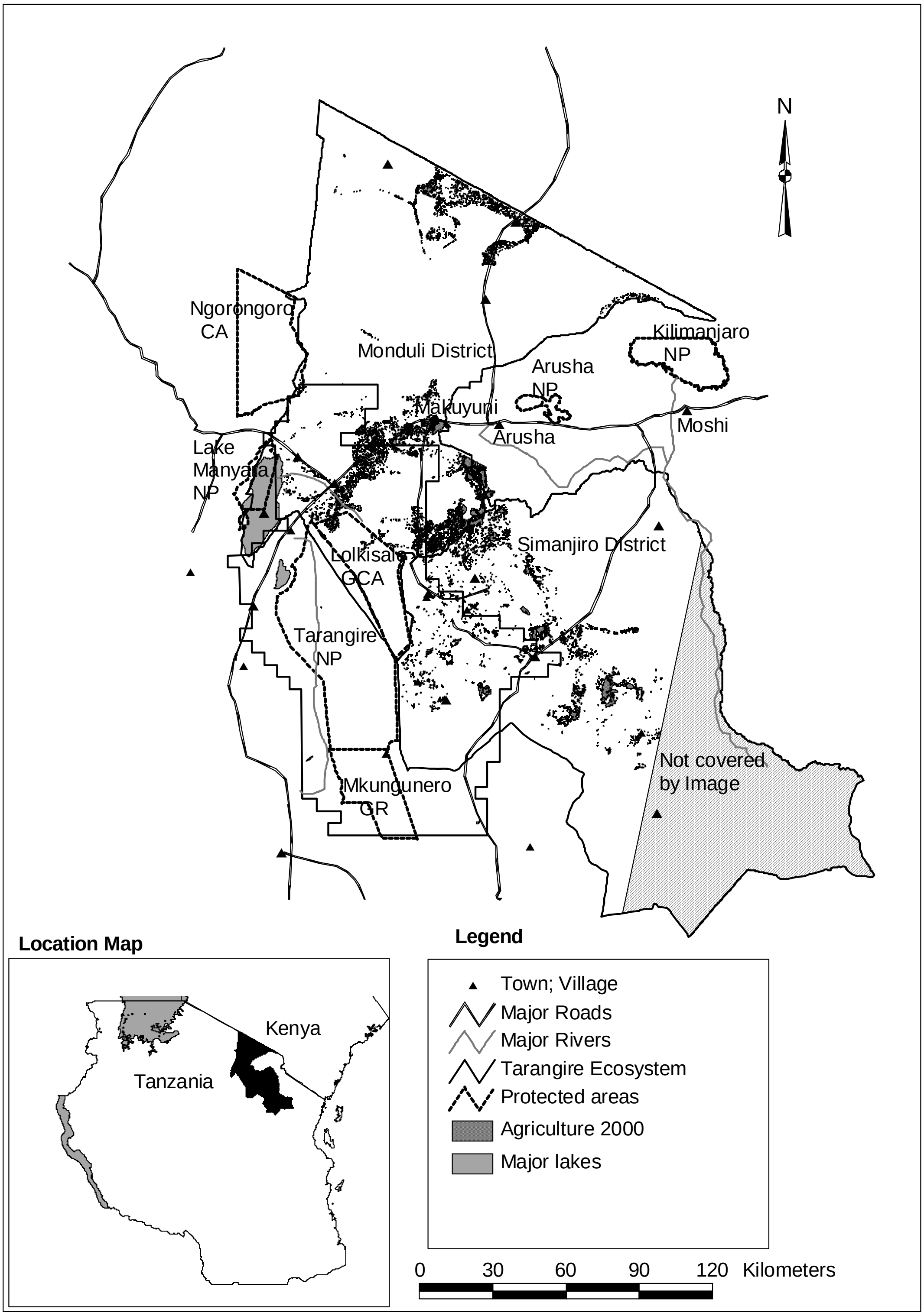

Figure 3.1 Map showing the study area and administrative boundaries and baseline conditions in 2000; main protected areas, town-villages, major roads, rivers and the agricultural expansion in the two districts of Simanjiro and Monduli, Northern Tanzania. NP = National Park; GR = Game Reserve; CA = Conservation Area; GCA = Game Controlled Area.

3.2.2 Data and methods

Spatial datasets were derived from remotely-sensed imagery, radio-collared animals and Geographic Information System (GIS) layers. The datasets were used to examine implications of land use changes and its drivers on wildlife habitats and distribution. The biophysical predictors of land use change were rainfall, slope, distances to the nearest village (town), road, river and protected area (parks) boundary. The GIS layers of the study area, protected areas, villages/towns, and data on the ranges of migratory wildlife derived from radio-collared animals were obtained from the GIS centre of the Tarangire National Park. The GIS layers for roads and rivers were acquired from the Surveying and Mapping Division of Tanzania based on 1:50,000 topographic maps. Elevation data were derived from a Digital Elevation Model (DEM) of 90 m resolution obtained from the Shuttle Radar Topographic Mission (SRTM). Data on rainfall (annual precipitation) were extracted from the Almanac Characterization Tool (ACTS) database (Mud Springs Geographers 2002).

Remote sensing data were extracted from satellite images, acquired from the USGS. The images were Landsat ETM+, Path/Row 168/62 and 168/63 of 2000 (Ref. Chapter 2, for the details of classification).

3.2.3 Statistical and spatial analyses

MLR model was used to evaluate the relative significance of factors influencing the probability of occurrence of agriculture in the study area. The MLR model was used to estimate coefficients of explanatory variables with the presence (1) or absence (0) of agriculture in each grid cell in the year 2000 as the dependent variable. Grids of 300 m × 300 m cells were generated, which were determined by the minimum parcel size of cultivated land. This grid was overlaid with a GIS layer for agriculture and assigned 1 to grid cells with agriculture and 0 otherwise.

The MLR model was fitted using restricted pseudo-likelihood in the SAS GLIMMIX procedure (SAS Institute, 2006). For each of the six variables; namely annual precipitation, slope, distances to the nearest village (town), road, river and park (protected area), regression analyses were performed using the linear, quadratic without a linear term and the standard quadratic model including a linear term. For each model, the Akaike Information Criteria (AIC) value and Akaike weights (Burnham & Anderson 2001) were computed. The AIC is an omnibus information-theoretic criterion that is widely used in model selection to achieve a trade off between predictive accuracy of a model and parsimony (Burnham & Anderson 2002). Models were then fitted with interaction terms and added and retained extra variables in the model only if this improved the value of the AIC. The procedure was repeated until all the explanatory variables had been considered.

Models were compared using ΔAIC, the difference between AIC for each individual model and the model with the lowest observed AIC value. Under this framework, the model with the smallest AIC value is interpreted as having the best fit to the data. Models with ΔAIC ≤2 suggests substantial evidence for the model, values between 3 and 7 indicate that the model has considerably less support, whereas ΔAIC >10 indicates that the model is very unlikely (Burnham & Anderson 2002). It was presumed that parameters with good support would have high Akaike weights (near 1) since that parameter would be included in most of the better models. Finally, the Goodness-of-fit test was used to assess the global model fit statistics (Anderson & Burnham 2002).

The Hawth's Analysis Tool (an extension for ARCGIS- ArcMap) was then used for the analysis of animal movements (range) from the radio-collaring data of key migratory species in the Tarangire ecosystem (http://www.spatialecology.com/htools/). The data were from 10 wildebeest and 13 zebra collared between 1995 and 1997 (OIKOS, 2002). Minimum Convex Polygons were first created to characterize the range of the radio-collared animals based on spatial locations derived from the Global Positioning System (GPS). The Batch Fixed Kernel Density Estimator was then used to derive a set of percentage volume contours/maps showing the intensities of habitat use within the range of the collared animals. To analyse the relationship between the range/habitat of the key migratory species and agricultural land-use, maps derived from the radio-collared animals (range-extent), their migratory routes and the spatial agriculture in 2000 in the study area were overlaid. The area of overlap between the range used by the key migratory wildlife species and the agricultural land-use was calculated to assess the extent of natural habitat lost to cultivation. Finally the migratory corridors and the key habitat/range of the species were overlaid to assess and classify the status of the corridors as open, threatened, or blocked.

3.3 Results

Comparisons of the individual variable models indicated that quadratic models were better supported than linear ones. Table 3.1 shows the 18 candidate models considered, their AIC, ΔAIC, Akaike weights and the rank order of the models. The standard quadratic models with precipitation (model 1), distance to the nearest town (model 4), park (model 7), road (model 10), river (model 13) and slope (model 16) had the highest support in the data. These models thus had the highest Akaike weights (100%) and were ranked as the best models. For the full model shown in Table 3.2, only significant interactions ($P < 0.05$) were retained.

3.3.1 Patterns of spatial distribution of cultivation and the biophysical variables

There was a humped distribution between the likelihood of agriculture and precipitation (Fig. 3.2). The probability of presence of agriculture increased significantly with increasing rainfall from around 300 mm to a peak ($p \approx 0.22$) around 800 mm of rainfall and declined with further increase in rainfall. The probability of presence of agriculture relative to terrain slope (Fig. 3.2) showed that cultivation was most likely to be practised in areas with slopes of about 10°. The probability of agriculture dropped consistently for areas with slopes lower or higher than 10° ($p \approx 0.18$).

Table 3.1 Information-theoretic model selection of a *priori* candidate models explaining the presence of cultivation in Simanjiro and Monduli Districts in 2000 and the bio-physical variables included in each model.

Number	Predictors in model	AIC value	AIC difference (ΔAIC)	Akaike weights (w_i)	Rank
1	annual precipitation, annual precipitation^2	227886	0	1	1
2	annual precipitation^2	233558	5672	0	3
3	annual precipitation	232287	4401	0	2
4	distance to town, distance to town^2	230178	0	1	1
5	distance to town^2	230180	1	0	2
6	distance to town	230909	731	0	3
7	distance to park, distance to park^2	240635	0	1	1
8	distance to park^2	240999	364	0	3
9	distance to park	240803	168	0	2
10	distance to road, distance to road^2	235409	0	1	1
11	distance to road^2	236431	1021	0	3
12	distance to road	235504	95	0	2
13	distance to river, distance to river^2	240149	0	1	1
14	distance to river^2	240357	207	0	2
15	distance to river	240723	573	0	3
16	slope in degrees, slope in degrees^2	239767	0	1	1
17	slope in degrees^2	241178	1411	0	2
18	slope in degrees	241186	1418	0	3

Table 3.2 Results of the multiple logistic regression of the probability of presence of cultivated farms against biophysical predictor variables for the Simanjiro and Monduli District of Northern Tanzania based on the Satellite remote sensing imagery for 2000. NDF and DDF are the numerator and denominator degrees of freedom for the F-test, respectively.

Effect	NDF	DDF	F	P> F
annual precipitation	1	320427	2731	<0.00001
annual precipitation × annual precipitation	1	320427	1952	<0.00001
slope in degrees	1	320427	644	<0.00001
slope in degrees × slope in degrees	1	320427	977	<0.00001
distance to road× distance to road	1	320427	1115	<0.00001
slope in degrees × distance to road	1	320427	73	<0.00001
distance to river × distance to river	1	320427	19	<0.00001
distance to town	1	320427	50	<0.00001
distance town × distance town	1	320427	415	<0.00001
distance to road × distance to town	1	320427	212	<0.00001
distance to river × distance to town	1	320427	3012	<0.00001
distance to park	1	320427	1682	<0.00001
distance to park × distance to park	1	320427	216	<0.00001
distance to river × distance to park	1	320427	160	<0.00001
distance to road × distance to park	1	320427	1392	<0.00001
distance to town × distance to park	1	320427	1914	<0.00001
slope in degrees × distance to road × distance to river	1	320427	108	<0.00001

The relationship between presence of agriculture and distance to the nearest village (town) declined with increasing distance from villages (Fig. 3.2). The probability for cultivation was highest ($p \approx 0.22$) within 0-10 km of villages. Similarly, the probability of finding agriculture declined with increasing distance from roads and was highest ($p \approx 0.17$) within 5-20 km from the nearest road, (Fig. 3.2).

The relationship between presence of agriculture and distance from the nearest river was concave and was highest ($p \approx 0.16$) nearest to (0-5 km) and farthest from (40-45 km) the nearest river and lowest at about 30 km from the nearest river (Fig 3.2). The probability of finding agriculture as a function of distance from the nearest park also showed a humped distribution with a non-zero probability for agriculture ($p \approx 0.08$) apparent up to 20 km inside the nearest protected area. The probability for cultivation initially increased with increasing distance up to about 30 km from the nearest protected area boundary and then declined steadily with further increase in distance (Fig 3.2).

When all the variables in the best univariate models were combined into a global multivariate model, it was apparent that the probability of finding agriculture was highest near rivers and towns, areas receiving high rainfall and near villages located near parks (Table 3.2). The probability of occurrence of agriculture in the Maasai-Steppe was significantly associated with areas of high agricultural potential irrespective of their protection status or importance as wildlife ranges.

The goodness-of-fit tests showed that the global model had good explanatory power. The Hosmer-Lemeshow test statistic (which includes three asymptotically equivalent Chi-Square tests, i.e. the Likelihood Ratio, Score and Wald) were highly significant ($p<0.0001$), supporting the fitted global model. The maximum-rescaled R-Square, (Nagelkerke 1991) was 0.19. The association of the predicted probabilities and the observed responses were 76.5% concordant and 23.1% discordant. The high level of concordance (agreement) is particularly important as it implies that the model reliably represents the processes underlying the patterns observed in this study (Hunter *et al.* 2003).

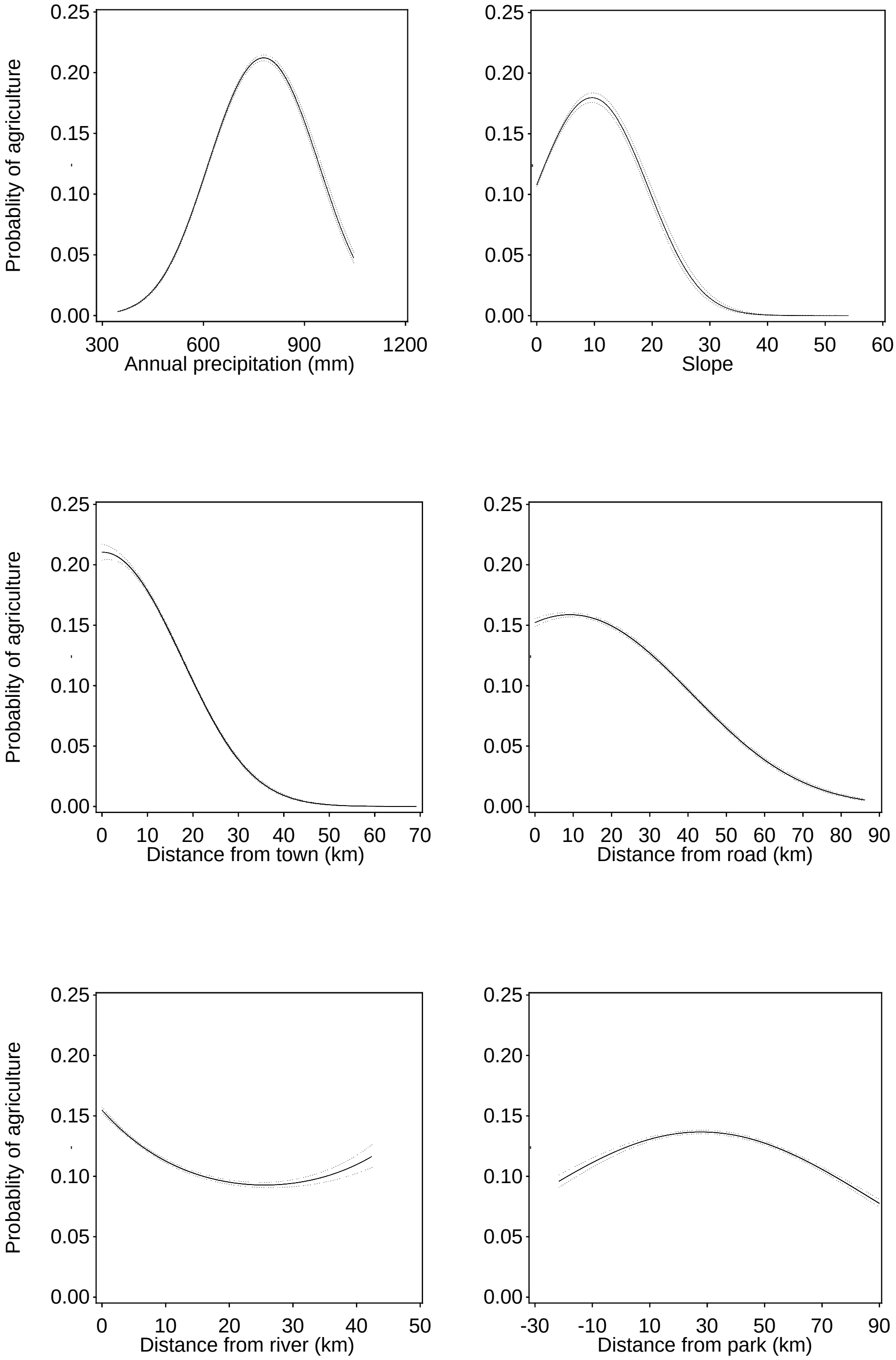

Figure 3.2 Relationship between the probability of presence of cultivation in 2000 and the biophysical variables in the Maasai-Steppe of Northern Tanzania.

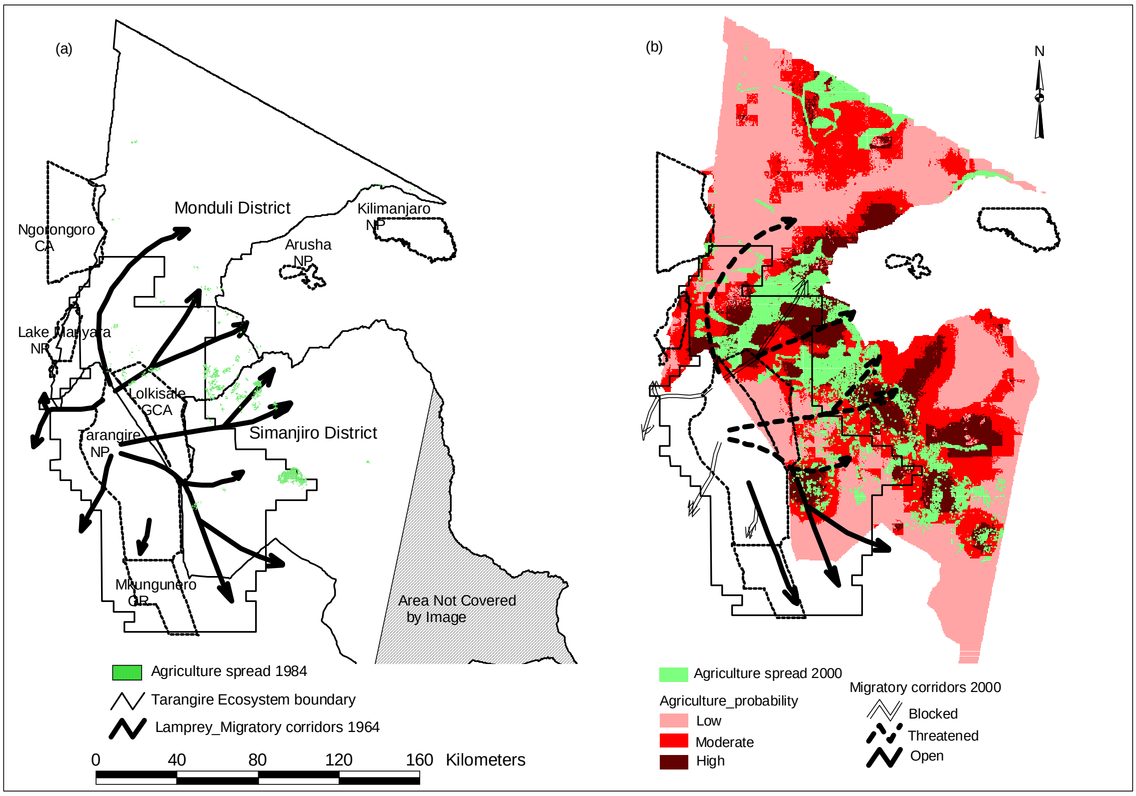

Figure 3.3 (a) Spatial distribution of agriculture in 1984 and wildlife migration corridors described by Lamprey in 1964, (b) Spatial distribution of agriculture in 2000 and the probability of further conversion to agriculture overlaid with the migratory wildlife corridors in the study area in 2000. (Source: Land-use data on agriculture for both 1984 and 2000 are based on remote sensing analysis from chapter II)

3.3.2 Mapping the probability of agricultural conversion and wildlife range

The probability map generated from the logistic regression model shows the relative likelihood of conversion to agriculture for the rangeland across the Maasai-Steppe

(Fig. 3.3.b). The map is based on the biophysical landscape variables which showed significant correlation with agricultural presence in 2000 and retained in the global multivariate model. The global multiple logistic regression equation used to predict the probability of cultivation in each pixel in the study area suggests how future expansion of agriculture in this region will be constrained by values of the biophysical variables analysed here. Further, the overlay of probability maps of agricultural occurrence and key migratory wildlife range and corridors from the radio-collaring data showed that approximately 13% of the range area for wildebeest and zebra had been converted to farms by 2000 (Fig. 3.4).

Four of the remaining five migratory wildlife corridors described by Lamprey in 1963 (Lamprey 1964, Fig. 3.3.a) are seriously threatened with blockage (Fig. 3.3.b). The first, in the north-east; the Kwakuchinja wildlife corridor, is used mainly by wildebeest and zebra from TNP to Manyara Ranch and Lake Manyara National Park. The second, the corridor from TNP through Lolkisale Game Controlled Area up to Losimingori Mountains, is used mainly by Elephants. The third corridor lying to the east and running from TNP to the Simanjiro Plains is used mainly by wildebeest and zebra moving to and from their calving grounds. And the fourth one, lying south-east from TNP to Loibor-Siret and Kimotorok villages is used by wildebeest, zebra and elephants. All these four corridors are currently being converted to extensive cultivation and settlements in the Tarangire ecosystem as indicated in Fig. 3.3.b

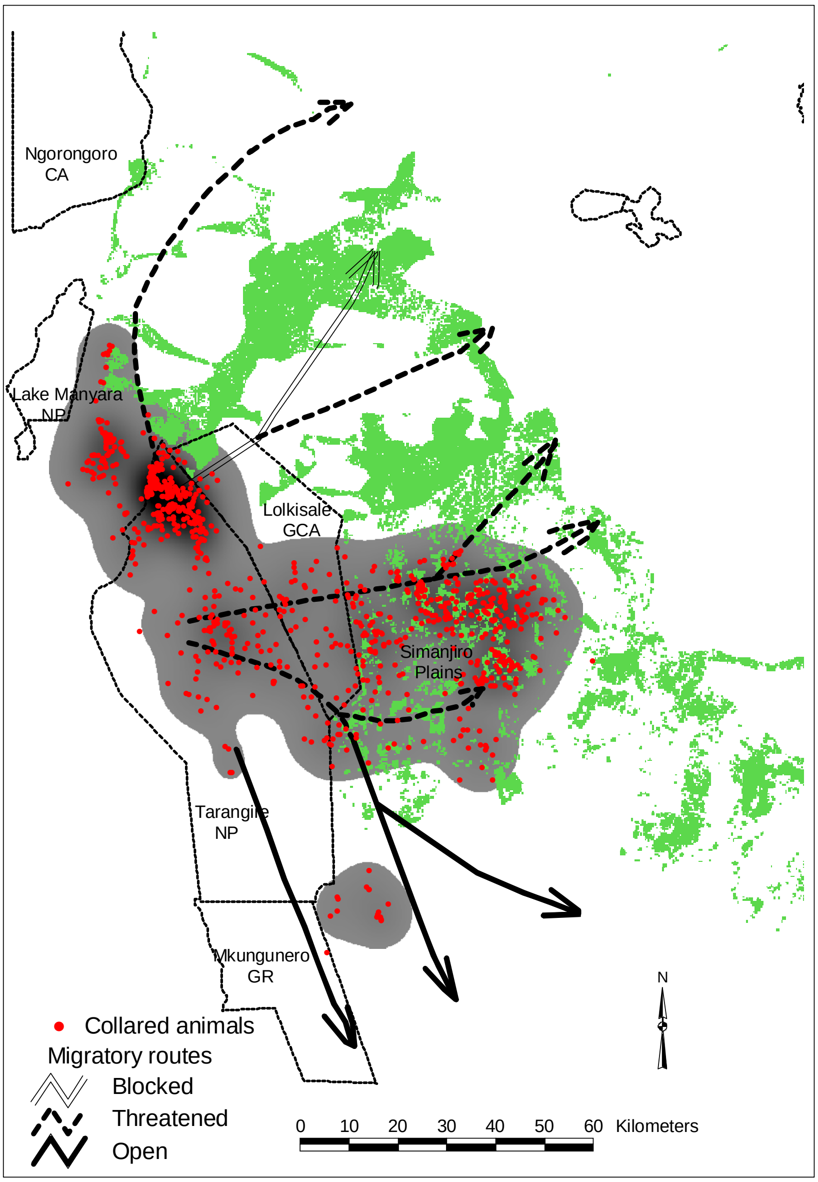

Figure 3.4 Wet season range of the two key migratory species in gray-tone based on data from radio-collared wildebeest (n=10) and zebra (n=13) overlaid with cultivation in 2000.

3.4 Discussion

The study shows the existence of a strong overlap between lands suitable for agriculture and the main wildlife corridors and the wet season dispersal areas. These results confirm and reinforce the earlier findings of Borner (1985) OIKOS (2002) and Msoffe et al. *(in press)* that agricultural encroachment is the single most important factor which blocked four of the nine wildlife corridors described by Lamprey, (1964) in the early 1960's. A new insight from this study is that four of the remaining five corridors also overlap with areas highly suitable for agriculture. The conversion to farming is occurring haphazardly leading to patchy and fragmented habitats, which cannot support many ecologically viable processes required by the migratory species. Clearly, expanding cultivation towards protected areas severely restricts wildlife movements to dispersal areas outside parks by blocking their corridors.

The assessment of potential for agriculture was based on two land suitability models. The first, a deductive biophysical land suitability model, considered rainfall and the topography, predicted a somewhat larger area suitable than the second model. The second model, an empirical model, considered, apart from rainfall and slope steepness, other factors such as distance to settlements and roads. The second model indicated that infrastructure played an additional important role. Biophysical variables provide the primary conditions that distinguish land suitable from that unsuitable for agriculture. Within this context, socio-political and economic drivers of land-associated decisions concerning where to develop infrastructure and prioritize which lands will be converted are made. The political dimension is clear when it is realized that infrastructure plays an overriding role, and its development is largely determined by governments at national and international levels. Indeed, roads do not emerge at random, but are mostly the result of deliberate development planning. Hence, governments would have the possibility to decide not to develop roads.

Similar observations have been made in other studies (Mertens & Lambin 1997,Serneels & Lambin 2001c,Jasinski *et al.* 2005,Etter *et al.* 2006). However, most of these studies found that access to roads and markets were much more important in relation to land-cover conversions to agriculture and deforestation. In contrast, this study showed that villages were, statistically speaking, more significantly related to agricultural presence/land-conversions than distance to roads. Villages were strategically located near rivers and water points where they also coincide with wildlife use of these areas particularly during the dry season. This unplanned land use is highlighted in this study by the diffuse distribution of agriculture. And, as more villages are settled, this problem will become greater and inimical to conservation endeavors and politically and economically much more costly to solve. Consequently, as more land is put under agriculture the range for both wildlife and livestock become increasingly diminished (FAO, 2009).

The diminishing range size implies that small stochastic events such as droughts could affect larger proportions of livestock and wildlife populations, especially so for mammals that directly threaten human lives, compete with humans for resources and/or are restricted inside artificial boundaries (Thuiller *et al.* 2006). Recent aerial surveys in the Tarangire-Ecosystem revealed extreme declines in numbers of the key migratory wildlife species, most notably wildebeest, whose numbers dropped from about 43,000 in 1988 to a mere 5,000 in 2001 (TAWIRI 2001). Long-term studies in other pastoral lands with large migratory populations in East Africa such as the Maasai Mara (Ottichilo *et al.* 2001,Ogutu *et al.* 2009), and Athi-Kaputiei (Reid *et al.* 2008) ecosystems of Kenya have also implicated agricultural expansion, loss of wet season dispersal ranges, expansion of settlements and urban development as primarily responsible for massive declines by populations of migratory wildebeest and other ungulates. Concerted efforts are being made in the Athi-Kaputiei and the Mara to develop innovative ways for keeping the land open for wildlife and pastoral livestock and consolidating small individual land parcels to form conservancies (Norton-Griffiths *et al.* 2008,Reid *et al.* 2008)

In conclusion, in this study remotely sensed data and statistics have been used to build models which allowed the prediction of where land-cover conversions are most likely to take place in the future and hence anticipate their associated impacts. Although some of these conversions have already occurred, more areas suitable for agriculture are still available that could soon be cultivated. Hence, governments may have contributed to the observed blockage of the remaining five migration routes because of its central role in road development and planning. Similarly, land tenure is a major driver affecting where settlements develop, and governments, through spatial planning and land tenure arrangements influence the location of settlements. It is questionable however whether such policy instruments would still be effective given the high political and economic costs of relocating large-scale settlements or cultivation.

Maps revealed that settlements in agricultural lands were dispersed all over the wildlife dispersal areas in 2000. Removing people from wildlife dispersal areas might be difficult once they have settled, as many of the settlers received official titles from the village government. However, land fragmentation has negative impacts on both wildlife conservation and tourism, one of the biggest revenue earners for the Tanzanian government (TNRF 2008). The government needs to develop strategies at both national and village levels that integrate the development needs of land, tourism, forestry and livestock sectors. It is imperative to integrate wildlife conservation needs with pastoral livestock production and broad development goals because wildlife and livestock both depend on the same resources and compete with farmers for land (TNRF 2008).

A major challenge for contemporary decision-makers is to develop forward-looking strategies that incorporate what is currently happening on the ground between neighboring villages and districts. Decision-makers also need to know what is happening where, what the causes of land use changes are and what alternative options are available in order to effectively plan land use, monitor impacts and learn and adjust the plans and strategy to meet their intended goals. Complete loss of wildlife dispersal areas and corridors will reduce protected areas to ecological islands where sustainable conservation of the species may not be possible even through active management strategies (Ottichilo *et al.* 2000,Newmark 2008). There is a need to pursue land-use plans that are both profitable and sustainable for communities but also compatible with wildlife conservation. The plans need to address the different land uses strategies in order to alter the observed trends and ease their cohabitation. It should limit the expansion of agriculture into key wildlife habitats, given the constraints of soil fertility and water in these semi-arid rangelands. More important, the plan should be able to support sustainable pastoralism and livestock which is the most productive use of these lands (FAO, 2009). And finally, the government should invest in and encourage use of simple methods of participatory land-use planning. When communities have accurate information on the pluses and minuses of farming, livestock keeping, wildlife, or other livelihood strategies they can best zone their land for different activities.

The process of modelling land development scenarios presented here demonstrates a potentially useful tool for policy makers, allowing for estimation and visualization of the land-use implications in conservation planning, land-use planning and policy decisions.

4 Chapter Four: Participatory wildlife surveys in communal lands: A case study from Simanjiro, Tanzania[i].

Abstract

It is widely accepted that protected areas alone are not sufficient to conserve wildlife populations particularly for migratory or wide-ranging species. This study assessed the population density of migratory species in the Tarangire-Simanjiro Ecosystem by conducting a ground census using DISTANCE sampling. The study focused on the Simanjiro Plains which are used as a dispersal area by wildebeest (*Connochaetes taurinus)* and zebra (*Equus burchellii*). This case study demonstrated that DISTANCE sampling can provide precise estimates of population density and is an affordable method for monitoring wildlife populations over time. Further the study underlines the importance of involving local communities in monitoring programs across landscapes that incorporate communal lands as well as protected areas.

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[i] Materials from this chapter have been published as: Msoffe, F.U., Ogutu, J.O., Kaaya, J., Bedelian, C., Said, M.Y., Kifugo, S.C., Reid, R.S., Neselle, M., van Gardingen, P. and Thirgood, S. (2009) Participatory wildlife surveys in communal lands. A case study from Simanjiro, Tanzania. *Afr. J. Ecol.*, **48**, 727-735

My contribution in this paper: I developed the concept in collaboration with my supervisory team, i.e. Reid, Ogutu, Thirgood, Said and van Gardingen. My supervisory team also supported in the field designs and I did the field preparations and the actual survey as a team leader with the close support of Ogutu, Neselle, Bedelian, Kaaya and Kifugo. My supervisory team also advised on the data analyses and commented on several versions of the draft manuscript until it was accepted for publication.

4.1 Introduction

There is a growing need for reliable monitoring of wildlife populations in the communal lands of Africa (Gaidet *et al.* 2005). Monitoring is a crucial source of information for determining conservation priorities and evaluating ecosystem responses to management activities, particularly in areas where wildlife is not formally protected (Georgiadis *et al.* 2003). Management can be more effective if based on a sound understanding of ecological processes including migratory movements and wildlife harvesting (Georgiadis, et al. 2003,Msoffe *et al.* 2007). Reliable estimation of population size represents an essential first step in monitoring the consequences of management activities.

Despite the fact that wildlife populations occur seasonally on communal lands adjacent to protected areas, most wildlife monitoring programs are conducted only in the protected areas (Nelson 2007,TNRF 2008). These communal lands are important for conservation as they provide seasonal foraging and calving areas for many species. The long-term conservation of these areas requires the participation of local communities in management. One way to ensure that communities are actively involved in conservation is to engage them in monitoring and resource protection activities so that they can generate benefits from having wildlife on their land (Baumgartner & Hartmann 2001,Gross 2007). Long-term monitoring of wildlife populations require precise and unbiased methods for estimating population size and related parameters so that temporal changes in these population characteristics can be reliably established.

In common with other communal land in Africa, the Maasai Steppe in Tanzania is threatened by anthropogenic factors affecting all aspects of biodiversity from species to communities (Borner 1985,Mwalyosi 1991a,OIKOS 2002). Within the Maasai Steppe, the Tarangire-Simanjiro Ecosystem (TSE) is one of the richest wildlife areas in Tanzania and is also an important tourism destination (Prins 1987). The TSE incorporates a number of protected and unprotected areas subject to different forms of natural resource use. At the heart of the ecosystem is the Tarangire National Park (TNP), which contains the only perennial source of water in the dry season. This coincides with the time of year when majority of large mammals, including the migratory herbivores, congregate along the Tarangire River (Lamprey 1963). At the onset of the rains, these animals disperse north and east away from TNP into the surrounding ecosystem (Lamprey 1964a). The factors driving these migrations are not fully understood but likely include mineral nutrition and the availability of green forage and surface water (McNaughton 1990,Kahurananga & Silkilwasha 1997a,TCP 1998,Gereta *et al.* 2004). Once outside the TNP, the migrants occupy unprotected areas where they co-exist with people, livestock and agriculture (Borner 1985,TAWIRI 2001,OIKOS 2002).

Large mammal censuses in Tanzania are usually conducted using aerial surveys, typically using the Systematic Reconnaissance Flight (SRF) technique (TWCM 2000). Population estimates from SRF surveys may have large confidence limits making it difficult to ascertain population trends (TAWIRI 2001). Moreover, only four aerial surveys have been conducted in the TSE during the wet season in the past 25 years. These aerial surveys were conducted as components of donor-funded projects. Without such external funding it is unusual for the Tanzania Wildlife Division (WD) to conduct aerial surveys incorporating communal lands because of the high costs involved. However, the WD sets hunting quotas each year in the TSE, and elsewhere in Tanzania. Setting hunting quotas without reliable information on population trends can endanger population viability especially if the species concerned is already threatened by illegal hunting (Msoffe *et al.* 2007).

Given the limitations of aerial survey, it is important to test alternative approaches for conducting wildlife censuses in the communal lands of Africa. One approach that is becoming increasingly popular for counting wildlife distributed across large areas is ground-based DISTANCE sampling (Buckland *et al.* 2001,Ogutu *et al.* 2006). Here the study tests the reliability of ground-based DISTANCE sampling for counting large mammals in the TSE. The objectives of the study were threefold: (1) develop and test a survey method that can be used to monitor wildlife populations outside protected areas; (2) obtain baseline estimates of wet season density for migratory species and livestock in the TSE; and (3) compare the costs and logistics of ground-based DISTANCE sampling with aerial survey.

a)

b)

Plate 3_ a) Elephants concentration in Tarangire National Park along the Tarangire River and b) migratory species such as zebra in the Simanjiro Plains during the wet season; note the presence of maize fields in this area

4.2 Methods

The study was conducted on the Simanjiro Plains in the TSE in Tanzania (Fig. 4.1). The survey focused on a 589 km^2 area located 40 km east of the TNP, which constitutes one of the wet season calving grounds for migratory wildebeest and zebra. DISTANCE sampling was used from ground surveys to count wildlife, livestock and people during the 2007 wet season (28th April to 12th May 2007). A team of 18 counters, comprising the team leader, six experienced counters, two district officials and eight community members, supported by one driver, conducted the counts based on the theory, assumptions and design considerations in Buckland *et al.,* (2001).

The survey team was trained for five days on DISTANCE sampling methodology. Distance sampling is a widely-used group of closely related methods for estimating the density and/or abundance of populations Buckland *et al.,* (2001). The main methods are line transects and point transect. This study employed the line transect method in which an observer records distances and angles (using laser range finders) of objects (animals in this case) from the centre of the transect line. The distances and angles are later on used in calculating the density and abundance of objects in question. Training focused on ensuring the following assumptions would be met: (1) objects on the centre of the line are detected with certainty so that the detection probability on the line is 1, i.e. $g\ (0) = 1$; (2) objects do not move towards or away from the transect line in response to the observer before distances are measured; and (3) distances from the centre line to each object are measured accurately. Training also covered the use of maps, geographical positioning system (GPS), laser range finders, binoculars, and data sheets.

The team was divided into five groups for the survey with each group comprising an observer, navigator and recorder. Each group had at least one community member and one experienced counter. The survey lasted for 10 days (3rd to 12th May 2007) and covered a total of 50 transects (25 transects each sampled twice) each of 5 km length. Transects were oriented North-South, following grid lines on 1:50,000 topographic maps and spaced a minimum of 1.5 km apart to minimize the probability of overlapping counts between transects, based on preliminary surveys (Fig. 4.1). The minimum separation distance of 1.5 km between transects ensured that the range finder, which was accurate to 1 km from the transect centre line, recorded only distances to groups of animals along the target transect and enabled comprehensive coverage of either side of the transect.

The survey employed walked line transects following Buckland *et al.,* (2001). Each group was given a map of the area with transects marked with GPS coordinates and overlaid with landmarks such as roads, village boundaries and habitat types. Each team used a GPS to navigate. The observer and recorder noted the date and time, sighting angles (using the GPS compass), sighting distance (using the laser range finder) and habitat characteristics of all observations of wildlife, livestock and people. Binoculars were used to assist in species identification. The counts started at ~0700h with each transect requiring three hours to complete and each team sampling one transect per day.

DISTANCE v5.3 (Thomas *et al.* 2006) was used to model detection functions and calculate estimates of density. Sighting distances and angles were transformed to perpendicular distances to the geometric centres of groups prior to analysis using the trigonometric relation: Perpendicular distance = $x \sin(\theta)$ with x = sighting distance (in m) and θ = sighting angle (in degrees). The perpendicular distances were right truncated at 600 m for wildebeest, zebra and cattle, 500 m for people and 400 m for gazelle (*Gazella granti*). Frequency histograms of perpendicular distances were plotted for each species and fitted models to the histogram based on the key function and series expansion approach in Buckland *et al.,* (2001). The models, including the uniform, half-normal and hazard rate key functions and associated series adjustments were fitted to the data for all species. Information-theoretic model selection, in particular the corrected Akaike Information Criterion (AICc), was used to select the detection function model with the best support in the data. The goodness-of-fit of the AICc-selected model was then assessed using Chi-square and Cramer von Misses tests and special attention paid to model fit close to the transect line, where a goodness of-fit test is crucial (Buckland *et al.* 2001). Model selection revealed that models allowing for observers as covariates had less support than models excluding observers.

The group size, encounter rate of groups, density and numerical abundance and the coefficients of variation for each species were estimated using DISTANCE v5.3. The logarithm of cluster size was regressed against the detection probability to correct for size-bias in cluster size as a function of sightability and adjusted the expected cluster size for sightability if the slope of the regression was significant at 0.15 (Buckland *et al.,* 2001). The percentage contribution of variation in each of the three components of density, namely group size, encounter rate and detection function, were examined to establish factors influencing the precision of the estimated population abundance (Ogutu *et al.* 2006). Transects were counted twice thus the estimated density was halved (Thomas *et al.* 2006).

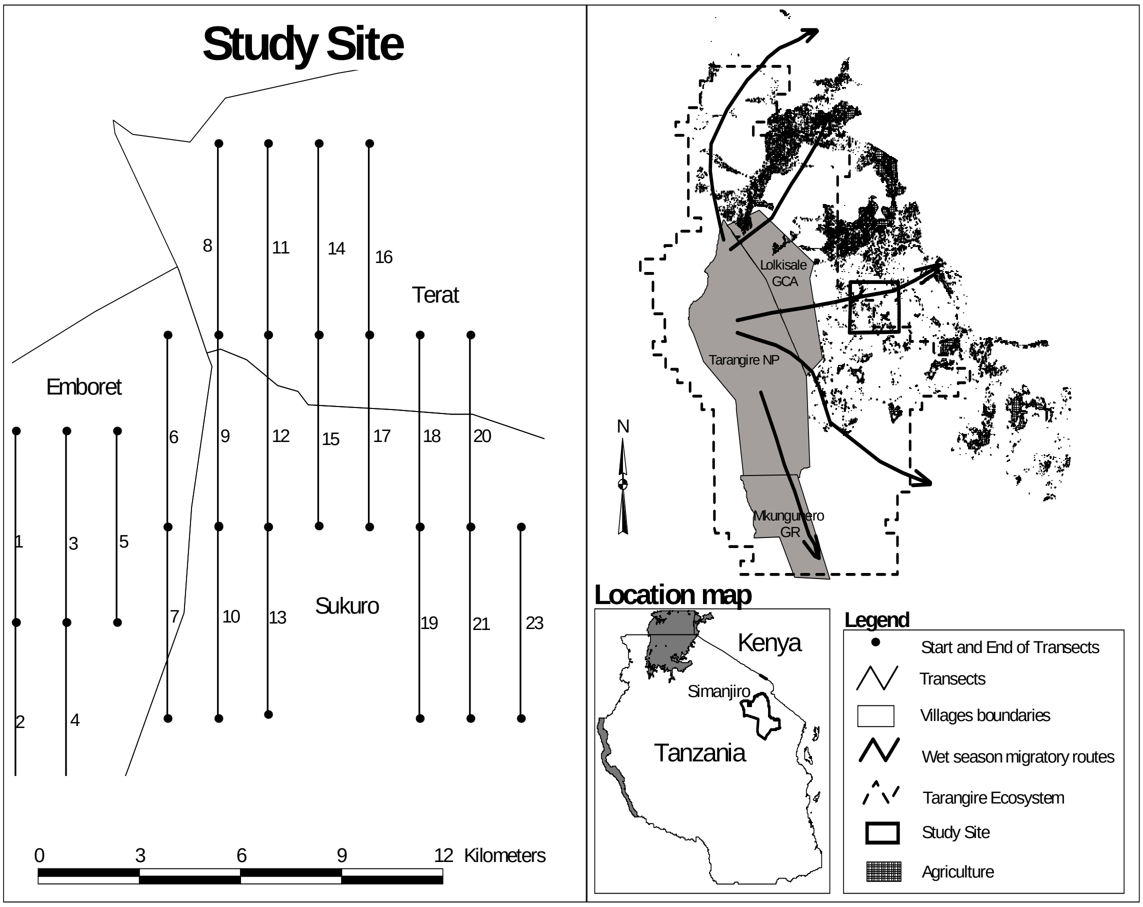

Figure 4.1 Map showing location of the study area in Northern Tanzania and the Tarangire-Simanjiro Ecosystem and the North-South transects layout in the three villages (Terat, Emboret and Sukuro) of the Simanjiro Plains.

4.3 Results

4.3.1 Survey implementation

The survey covered 589 km^2 and the distance walked on transects was 250 km. The time taken on transects was 150 h with an additional 40 h required for the initial period of training. These distances and times do not include the distance and time spent travelling from the field camp to the transects. The cost of implementing the survey (15 days including 5 days for training and 10 days of counting for a team of 18 people) was USD 10,000 or USD 40 km^{-1}. The costs were distributed as follows: salaries and per-diems for participants USD 4500; food and accommodation USD 3600; vehicle hire and fuel USD 1350; vehicle maintenance USD 150; GPS and range-finder batteries USD 200; and other field equipment USD 200.

The cost of the current ground survey can be compared to aerial surveys. Systematic Reconnaissance Flight (SRF) surveys using a Cessna 182 fixed-wing aircraft cost USD 11 km^{-1} inclusive of all aircraft and staff costs (Honori Maliti, pers. com.). Aerial surveys are typically flown over large areas and costs are therefore not strictly comparable to the current ground survey. DISTANCE sampling techniques can also be applied over smaller areas using helicopters but the costs of such surveys average USD 70 km^{-1}, and are higher than the costs for fixed-wing aircraft surveys (Jachmann, 2002).

4.3.2 Density Estimates

Twenty species were observed during the survey but only five were sufficiently abundant to estimate using DISTANCE. The five most abundant species were wildebeest, gazelle, zebra, cattle and people. Based on AICc and the chi-square goodness-of-fit tests, the half-normal key function with one cosine adjustment term was selected as the best approximate model for the detection functions for wildebeest, zebra and people. The uniform key function with one cosine adjustment term was selected as the best model for gazelle and cattle. The uniform key function without any series expansion was used with the wildebeest data and the same right-truncation distance of 600 m as used for the line transect to obtain a strip-transect estimate of wildebeest density for comparison with the line-transect method (Table 4.1). Precision in line-transect estimates of abundance is influenced by the variance in group size, encounter rate of groups and the effective strip width (Ogutu *et al.* 2006).

Plots of theoretical detection functions fitted to the observed frequency histograms of perpendicular distances showed that the probability of detection declined rapidly with increasing distance from the transect line at around 100-150 m (Figs. 4.2.a-e).

Table 4.1 The selected key functions and series expansions (model) for the detection function based on AICc values for each of the five species analyzed by program DISTANCE. S-poly means simple polynomial while H-poly means hermite polynomial; ** means the likelihood optimization algorithm failed to converge to an optimal value hence parameter estimates of models and fit statistics could not be computed**

Key	Series expansion	Wildebeest	Grant's gazelle	Zebra	Cattle	People	Wildebeest[1]
Uniform							2584.36
	Cosine	2499.36	729.51	536.75	444.38	839.28	
	S-poly	2499.36	729.51	****	445.24	****	
	H-poly	*****	****	536.89	446.99	****	
Half-normal	Cosine	2498.86	729.68	536.15	444.43	837.65	
	S-poly	2498.88	729.68	536.15	444.43	****	
	H-poly	2498.88	****	****	444.43	****	
Hazard-rate	Cosine	2500.80	****	****	447.21	****	
	S-poly	2500.80	****	****	****	****	
	H-poly	2500.80	****	****	****	****	

[1] The same data for wildebeest collected using line-transect was re-analyzed here using strip-transect method for comparison.

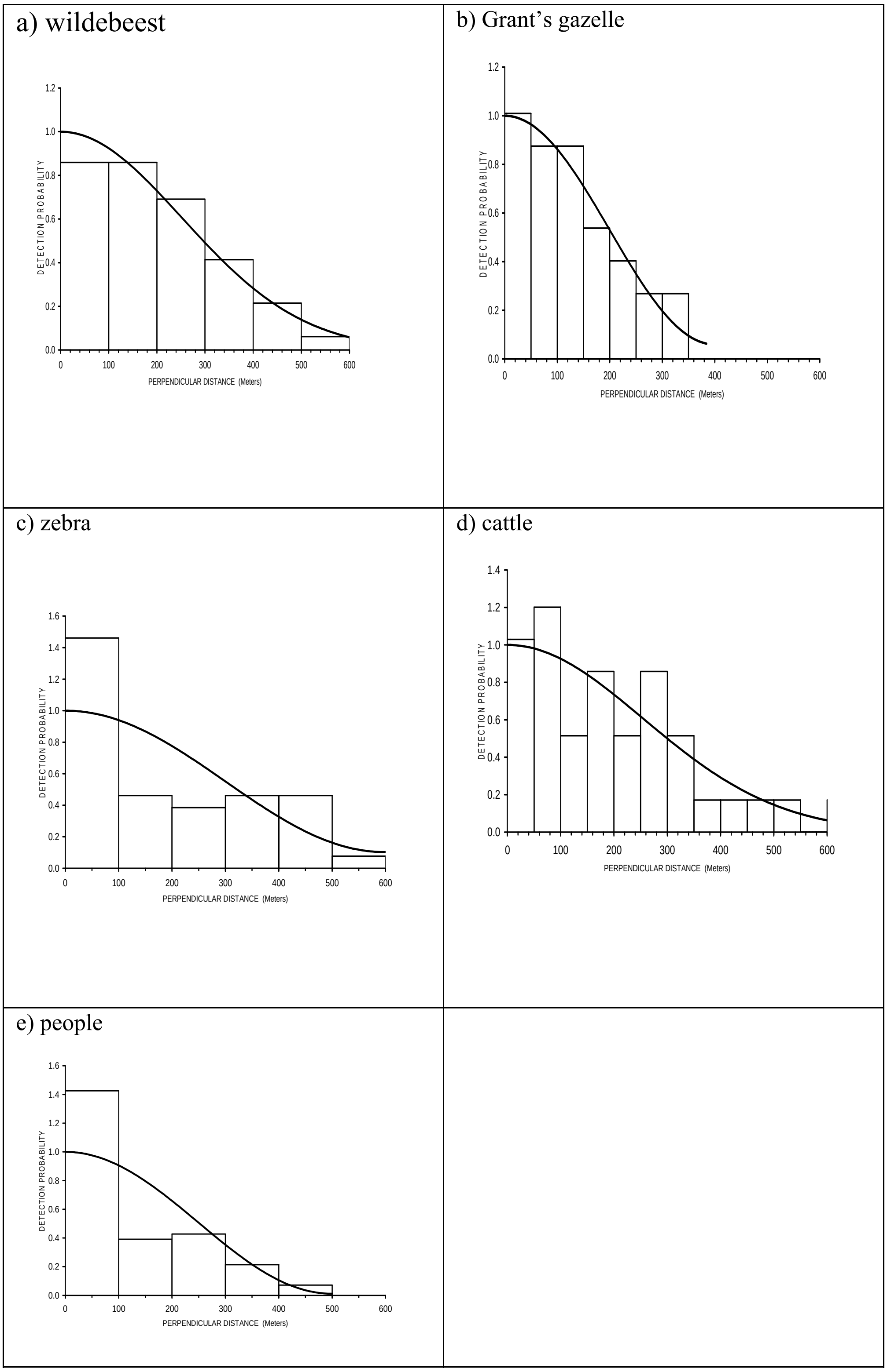

Figure 4.2 The observed frequency distributions of perpendicular distances (in meters) and fitted detection probability for wildebeest, Grant's gazelle, zebra, cattle and people.

Table 4.2 presents the results of estimates of abundance for the five species from DISTANCE and compares precisions of line-transect and strip-transect estimates of wildebeest density. Although abundance and density estimates for wildebeest were larger under the strip-transect than the line-transect method, estimates for the latter method were more precise (compare D-CI and D-CV in Table 4.2). The strip-transect estimate of density was based on a strip half-width of 600 m whereas the line-transect estimate was based on an effective strip half-width of 310 m, resulting in more groups being included in the strip-transect than the line-transect analysis. This could partly account for the slight upward bias in the estimated strip-transect density relative to the line-transect density. The mean group sizes were the largest for cattle and wildebeest followed by zebra, gazelle and people (Figs 4.3.a-e). Conversely, the encounter rates were higher for cattle, followed by zebra, people, gazelle and wildebeest. Variance in encounter rates made a greater contribution to variance in the estimated density for all species but wildebeest. Variance in group size made the next highest contribution to the variance in density whereas variation in the detection probability made the least contribution (Table 4.3).

Table 4.2 The observed number of clusters (n), estimated density ($\hat{D}$) and abundance ($\hat{N}$) and the 95% confidence limits (D-CI) and percent coefficient of variation (%CV) for people, cattle and the three common species of wildlife counted in the Simanjiro plains from 3rd to 12th May, 2007. ESW is the estimated strip half-width on either side of the transect centerline in metres.

Species	Observations (n)	Density ($\hat{D}$)	Abundance ($\hat{N}$)	D-CI (95%)	%CV	Cluster size (Es)	ESW
Wildebeest	202	9.4	5,531.9	5.8 – 15.3	20	14.5	310.1
Grant's gazelle	63	2.0	1,177.0	1.4 – 3.0	20	6.9	211.9
Zebra	43	1.7	1,000.5	0.9 – 3.2	30	12.9	324.1
Cattle	36	16.9	9,945.7	9.6 – 29.7	30	145.1	309.1
People	71	0.8	470.8	0.5 – 1.4	30	2.5	222.9
Wildebeest[1]	202	10.7	6,297.0	5.8 – 19.7	30	31.8	600.0

Table 4.3 Component percentages of the variance of density [var ($\mathrm{var}(\hat{D})$]: Cluster size, encounter rate and detection probability and their percent coefficients of variation (%CV)

Species	Cluster size (E (s))	% CV	Encounter rate (n/L)	% CV	Detection probability ($\hat{f}_0$)	% CV
Wildebeest	62.2	19.1	32.1	14.1	15.7	6.0
Grant's gazelle	34.8	11.1	53.4	13.8	11.8	6.5
Zebra	23.2	15.3	62.2	25.1	14.6	12.2
Cattle	30.9	16.1	63.7	23.2	5.4	6.7
People	36.6	17.3	54.5	21.1	8.9	8.5

4.4 Discussion

This study has demonstrated that it is possible to design and implement a ground survey using DISTANCE sampling to obtain reliable estimates of density for migratory ungulates and livestock in the communal lands of the TSE. Here the financial and logistic implications of the survey are evaluated, the precision of the density estimates derived from DISTANCE sampling compared to aerial surveys, and the extent to which local communities can be active participants in such surveys.

The cost of the current survey was USD 10,000 which equates to USD 40 km^{-1} of transect walked. A financial comparison suggests that DISTANCE sampling using ground surveys is four times more expensive per km than the costs of aerial survey using SRF techniques. However, such a simple comparison is misleading for four reasons. First, aerial surveys typically have a low intensity of spatial coverage making it difficult to reliably estimate density for small but important wildlife habitats. Second, aerial surveys strictly require DISTANCE sampling techniques to correct for sighting bias, which when implemented with a helicopter, which can fly slowly and at lower heights to facilitate reliable observation and distance measurements, cost six times more than a fixed-wing aircraft. This makes aerial DISTANCE sampling prohibitively expensive in most circumstances (Jachman, 2002). Third, aerial surveys require the use of aircraft with specialized personnel and equipment which are often unavailable for communal lands. Such surveys also preclude participation by community stakeholders in monitoring programmes. Finally, important ancillary information such as age and sex structure can be obtained during ground-based but not aerial surveys. It follows that the development and evaluation of ground-based survey methods that require inexpensive low technology equipment and allow the widest participation of community stakeholders could enhance conservation goals on communal lands.

Ground-based wildlife surveys in Tanzania have traditionally adopted strip-transect sampling rather than line-transect sampling. The financial costs of implementing strip-transect surveys are identical to those for line-transect surveys. Strip-transect and line-transect surveys are special cases of DISTANCE sampling and both require estimation of the perpendicular distance of animals from the transect centerline. However, while line-transects account for changing detection of animals with covariates, strip-transects assume that all animals within the strip are sighted with certainty. This assumption is violated under most field conditions, with observed sighting probabilities declining with distance, as confirmed for all species in our study.

The analyses revealed that precision in density estimation for DISTANCE sampling was more sensitive to variation in encounter rate than in group size and sighting probability. Although reducing the variance in encounter rate in order to increase the precision of the estimated density is desirable, this can be hard to achieve if animals are intrinsically highly clumped in their distribution. Achieving the minimum recommended sample size of 60-80 groups required to reliably estimate density with DISTANCE sampling in a single ground survey may prove difficult for many ungulate populations (Ogutu *et al.,* 2006). This problem can be overcome by pooling data from repeat surveys to estimate a common detection function and then using the pooled detection function to obtain density estimates for the different surveys. For long-term population monitoring this introduces no additional financial and logistic costs but ensures unbiased estimates of density relative to strip-transects.

One of the advantages of DISTANCE sampling is that it allows for the widespread observation that detection decreases with increasing distance from the transect line (Buckland *et al.* 2001). Assuming a constant sighting probability is unrealistic even for landscapes as open as the Simanjiro Plains. This partly explains why the density estimate for wildebeest based on the strip-transect method was biased relative to the line-transect estimate. The strip-transect estimate of density was also less precise than the line-transect estimate, as has been found in other studies (Newey *et al.* 2003,Bardsen & Fox 2006).

Estimates of population abundance for wildebeest, gazelle and zebra derived from our DISTANCE sampling survey were an order of magnitude more precise than estimates derived from SRF aerial surveys. For example, the 2001 Wet Season SRF survey counted 309 wildebeest on the Simanjiro Plains resulting in a population estimate of 5257 with standard error of 2616 and 95% confidence interval of 5-10,489 (TAWIRI 2001). Whilst there is superficial concordance between the population estimates derived from the two surveys, the coefficient of variation of the SRF estimate was 201% which compares unfavourably to the coefficient of variation of the DISTANCE estimate of 20%. Population estimates derived from the SRF aerial survey for gazelle and zebra also had very large coefficients of variation and confidence intervals (TAWIRI 2001). The estimated densities from the DISTANCE survey can therefore provide more reliable baseline data to assess the population status of key species in the TSE than estimates from the aerial surveys. In addition, because DISTANCE sampling on walked transects can focus on small discreet areas, it is possible to conduct censuses in hunting blocks and communal areas to provide valuable information to guide management activities (Waltert *et al.* 2006).

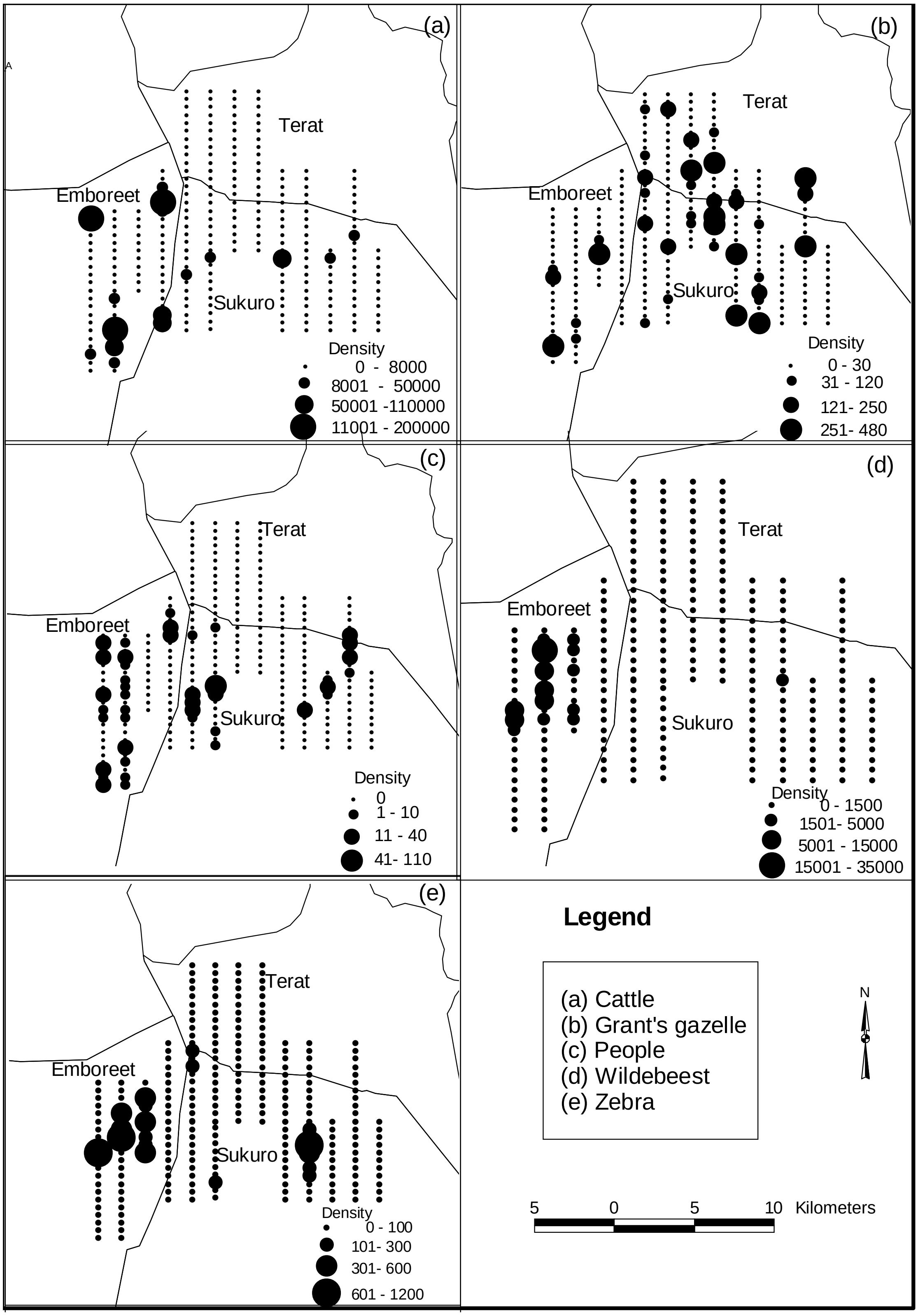

Figure 4.3 Spatial distribution of the observed groups for (a) Cattle (b) Grant's gazelle (c) People (d) Wildebeest and (e) Zebra based on DISTANCE sampling count in the Simanjiro Plains from 3rd to 12th May 2007.

Reliable and cost-effective monitoring of wildlife populations is becoming increasingly important in the development of community-based natural resource management (CBNRM) in Tanzania (Nelson & Ole Makko 2003,Kibebe 2005). There is also increased interest and debate concerning the decentralization of management of wildlife resources (Gaidet-Drapier *et al.* 2006,Nelson 2007). The move towards CBNRM will require wildlife enterprises such as community-based organizations (CBOs), tourism operators and hunting companies to accurately monitor wildlife populations to be able to set sustainable hunting quotas and monitor trends of key species over time.

This study has demonstrated that ground surveys based on DISTANCE sampling can provide accurate estimates of density for key wildlife species in communal lands. It is also clear that aerial surveys using SRF techniques often produce estimates of abundance which are too imprecise to reliably establish population trends. With a small investment of time and money, members of local communities can be trained to collect DISTANCE data with acceptable levels of accuracy. With support and encouragement, community members can plan and implement ground-based surveys on communal lands where it is logistically and financially unfeasible to conduct aerial surveys. External technical advisors will need to supervise data collection and analysis until such activities are within the skills of local communities. As such, the present survey represents an intermediate level of community involvement in natural resource monitoring where local people participate in monitoring activities but require technical input to interpret the resultant data (Danielsen *et al.* 2008). This active participation in monitoring activities helps to develop support from community stakeholders for CBNRM programs.

5 Chapter Five: Wildebeest Migration in East Africa: Status, threats and conservation measures

Summary

Migration of ungulates is under pressure worldwide from range restriction, habitat loss and degradation, anthropogenic barriers and poaching. This chapter synthesizes and compares the extent of historical migrations of the blue wildebeest (*Connochaetes taurinus*), in five ecosystems in East Africa. Records from colonial maps, literature reviews, GIS databases and interviews with local and resident researchers were used to analyse the current status of migration, migratory ranges and corridors. Interference of migratory corridors and dispersal ranges of the species have stopped and/or threaten the continual patterns of movements in all but the Serengeti-Mara ecosystem. Land use change also here use singular from agricultural encroachment and fencing have denied the species access to their grazing resources in unprotected lands. Land tenure change from group ranches to private ownership in Kenya and settlement policy (Villagization) in Tanzania which resulted in land subdivisions, fencing and permanent settlements have led to loss of key habitats for the species including their migratory corridors/routes, calving grounds and grazing areas during the wet season. Under the current policy environment, increasing human population pressure and climatic variability in these ecosystems, the study proposes urgent interventions by governments for conserving one of the Earth's remaining and most spectacular migrations.

5.1 Introduction

Ungulate migrations are among the most-awe inspiring of all migrations (Bolger *et al.* 2008). Migration, the seasonal and round-trip movement of large herbivores between discrete areas is under increasing pressure worldwide. Six of 24 species of ungulates which once migrated globally are already either extinct or the status of their migration is unknown (Harris *et al.* 2009). Of the remaining ungulate migrations, most occur in six locations in Africa, including the wildebeest (*Connochaetes taurinus*) migration in the Serengeti-Mara ecosystem of Kenya and Tanzania and the white eared-kob (*Kobus kob*) migration in Sudan (Harris, *et al.* 2009). Range restriction, habitat alteration, degradation and loss due to agriculture, poaching and anthropogenic barriers, such as fences, roads, railroads, pipelines and settlements are considered to have progressively disrupted historical migratory routes and driven many of the once spectacular migratory herds to extinction over the past century (Berger 2004,Reid *et al.* 2008,Bolger *et al.* 2008).

The preservation of the phenomenon of migration requires conservation of their habitats, a sound understanding of the factors and processes underlying the degradation and loss of migratory routes and declines of populations to devise effective strategies for protecting migratory routes, habitats and populations (Harris *et al.* 2009). Although causes of ungulate migrations are not yet fully understood (Sinclair 1995), the temporal regularity of migrations suggests that they are a response to seasonal fluctuations in spatial patterns of resource availability and quality. Thus rainfall through its effect on food supply and salinity of drinking surface water has been suggested as trigger of the northward migration (Wolanski *et al.* 1999) while high nutrient availability on the short grass plains is thought to attract lactating female wildebeest southwards (McNaughton 1990) in the Serengeti-Mara ecosystem. This migration results in a movement of wildebeest from the open grasslands, with low biomass in the wet season, to wooded grasslands, with high biomass during the dry season (Fryxell & Sinclair 1988).

The interplay of multiple factors and processes underpins threats facing ungulate migrations worldwide (Thirgood *et al.* 2004,Berger 2004,Bolger *et al.* 2008,Harris *et al.* 2009). The kind and intensity of these factors and processes vary among species and for each species among meta-populations. Consequently, it is useful to review trends and threats across a broad range of meta-populations of a migratory species to distill generic insights on threats they face and identify approaches likely to succeed in conserving their populations and migrations. This study describes and compares the extent of historical migrations of the blue wildebeest or brindled gnu, *Connochaetes taurinus* (Burchell, 1823), the current status of the migrations and migratory routes in five ecosystems of East Africa. Long-term wildebeest population trends, descriptions of the drivers of change and their impacts on the critical habitat and migratory ranges of wildebeest in each of the five ecosystems are examined and discussed and potential strategies for conserving the Earth's remaining most spectacular migrations are suggested.

Plate 5.1. Wildebeest migration in the great Serengeti-Mara ecosystem in East Africa.

5.2 Materials and methods

5.2.1 Study Area

This study covers the five ecosystems in East Africa with migratory wildebeest populations (Fig. 5.1). The first is the Serengeti-Mara ecosystem in Kenya and Tanzania, covers approximately 25, 000 km^2 (Thirgood *et al.* 2004). The southern migration in this ecosystem is the most extensively documented and involves about 1.3 million wildebeest, 0.2 million zebra (*Equus burchelli*) and 0.4 million Thomson's gazelle (*Gazella thomsonii*) (Talbot & Talbot 1963,Thirgood *et al.* 2004). The migration occurs mostly within protected areas, notably the Serengeti National Park, Ngorongoro Conservation Area, Maswa, Grumeti and Ikorongo Game Reserves and Ikoma Wildlife Management Areas in Tanzania, and the Masai Mara National Reserve and adjoining pastoral ranches in Kenya. A second migration in this ecosystem, the northern migration of the Mara-Loita-Siana population, occurs entirely in Kenya between the Masai Mara National Reserve and the Loita Plains to the north east through the Siana, Olkinyei and Koyiaki pastoral ranches (Stelfox *et al.* 1986,Homewood *et al.* 2001,Ottichilo *et al.* 2001,Serneels & Lambin 2001b).

The third wildebeest population migrates in the Athi-Kaputiei ecosystem, of Kenya covering the Nairobi National Park, and the adjacent Athi-Kaputiei Plains, an area of approximately 2,500 km^2. Wildebeest use the park during the dry season due to its better water supply and abundant grass and move onto and calve in the pastoral lands during the wet season (Forster & Kearney 1967,Forster & Coe 1968,Gichohi 1996).

The fourth wildebeest population migrates in the Amboseli ecosystem of Kenya covering the Amboseli National Park and surrounding dispersal areas on pastoral rangelands, an area of approximately 3,000 km^2 (Western 1975). The Park is a dry season refuge and herds disperse to the adjacent pastoral ranches in the wet season (Western 1975,Western *et al.* 2009a).

The fifth East African wildebeest population occurs in the Tarangire ecosystem of Tanzania, covering the Tarangire National Park forming the dry season range for the migratory herds, the Simanjiro Plains forming the wet season dispersal area and calving grounds, while the Mkungunero Game Reserve, Lolkisale Game Controlled Area, Manyara Ranch, Lake Manyara National Park and adjacent Game controlled areas (used mainly as hunting blocks), form other dispersal areas for the species, altogether covering an area approximately 35,000 km^2 (Lamprey 1964,Borner 1985,Kahurananga & Silkilwasha 1997,OIKOS 2002).

5.2.2 Mapping migratory routes and ranges

Information on the migratory wildebeest range, corridor/routes and their status was compiled from several sources, including literature reviews, colonial records, maps, GIS databases, and interviews with local residents and researchers knowledgeable about these ecosystems. Table 5.1 summarizes the information collated for each of the five migratory populations in Kenya and Tanzania.

Table 5.1 The five ecosystems with migratory wildebeest population in East Africa, the threats they face and the current status of the migrations.

Ecosystem	Threats and status of migration	Source/reference
Serengeti-Mara ecosystem (Kenya & Tanzania)	Still intact (14, 15 & 16); threats on the Kenyan side from large scale cultivation in the Mara(23, 24, 25) influencing the discharge of the Mara River impacting drinking water availability inside the ecosystem (13) and poaching in the Western Serengeti in Tanzania by an increasing human population (12, 14)	Pearsal (1957)[1]; Swynnerton (1958) [2]; Grizmek & Grizmek (1960) [3]; Talbot & Talbot (1963) [4]; Sinclair (1973) [5]; Pennycuick (1975) [6]; Kreulen (1975) [7]; Sinclair & Griffiths (1979) [8]; Maddock (1979) [9]; Fryxell & Sinclair (1988) [10]; Murray (1995) [11]; Mduma, et al. (1999) [12]; Gereta et al. (2003) [13];Thirgood et al. (2004) [14]; Musiega & Kazadi (2004) [15]; Boone etal. (2006) [16]; Haldo et al. (2009) [17]
Mara-Loita-Siana ecosystem (Kenya)	Highly threatened (23), migrating herds highly reduced due to land subdivision and privatization (22, 25, 26), expansion of cultivation (24), settlements, human population growth, sedentarization of formerly semi-nomadic Maasai pastoralists, intensification of land use and illegal human harvests (26, 27)	Darling (1960) [18]; Stelfox et al. (1986) [19]; Broten & Said (1995) [20]; Grunblatt et al. (1996) [21]; Norton-Griffiths (1996) [22]; Ottichilo et al. (2001) [23]; Serneels & Lambin (2001) [24]; Homewood et al. (2001) [25]; Norton-Griffiths (2008) [26]; Ogutu, et al. (2009) [27]

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Ecosystem	Threats and status of migration	Source/reference
Athi-Kaputiei ecosystem (Kenya)	Highly threatened by human population growth, land subdivision and privatization, expansion of settlements, fences, mines, quarries, dams, roads, urbanization, spiraling water extraction, commercial flowers, tree plantations, commercial charcoal burning (33, 34, 35)	Talbot & Talbot (1963); Forster & Kearney (1967) [28]; Forster & Coe (1968) [29]; Forster, J.B. & McLaughlin, R. (1968)[30]; Owaga (1975) [31]; Hillman & Hillman (1977) [32]; Gichohi (1996) [33]; Gichohi (2000) [34]; Reid et al. (2008) [35]; Ogutu et al. (in Prep.)
Amboseli ecosystem (Kenya)	Threatened by land subdivisions (37), fragmentation, fencing; human population growth, expansion of settlements and irrigated cultivation in Amboseli swamps (39, 40)	Western (1975) [36]; Western (1982) [37]; Campbell et al. (2000) [38]; Western (2007) [39]; Western, Groom & Worden (2009) [40]
Tarangire ecosystem (Tanzania)	Highly threatened by large-scale and subsistence cultivation; human population growth, blockage of migratory routes, loss of calving grounds and wet season habitats (42, 43, 44, 46 &47)	Lamprey (1964) [41]; Borner (1985) [42]; Kahurananga & Silkulwasha (1997) [43]; TAWIRI (2001) [44]; Oikos (2002) [45]; Gereta et al. (2004) [46] and Msoffe et al. (in Press) [47]

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5.2.3 Wildebeest population trend

Wildebeest population estimates were compiled from aerial surveys conducted in Kenya by the Department of Resource Surveys and Remote Sensing (DRSRS) and in Tanzania by the Tanzania Wildlife Research Institute (TAWIRI), Tanzanian Wildlife Conservation Monitoring Unit (TWCM) and Frankfurt Zoological Society (FZS). The methods used in the aerial surveys are described in detail elsewhere (Norton-Griffiths 1978,Grunblatt *et al.* 1996,Woodworth & Farm 1996). Aerial surveys began in the Serengeti-Mara ecosystem in the late 1950s (Talbot & Talbot 1963) and in the other ecosystems around the late 1970s.

5.2.4 Distribution of cultivation and fences

Data on the distribution of agriculture were obtained from the FAO, Africover project (2000). The project mapped land cover for the year 2000 for the whole of East Africa from Landsat images (30 m resolution). The map category agriculture was extracted from the Africover data set and clipped according to the study area boundary. In the Athi-Kaputiei ecosystem fences were mapped in 2004 and 2009 by the International Livestock Research Institute (ILRI) in collaboration with the local communities and local NGO's using hand-held Global Positioning System (GPS), with scientific, technical and logistical support provided by ILRI.

5.2.5 Statistical Analysis

Estimates of wildebeest population sizes, obtained using Jolly's method II for unequal transects (Jolly, 1969), were log transformed and related to the year of survey using linear and quadratic polynomial regression models and serial autocorrelation in the counts accounted for using the first-order autoregressive model. Model selection was based on the Akaike Information Criterion (Burnham & Anderson 2002). The models were fitted in SAS (SAS Institute, 2009).

5.3 Results

5.3.1 Historic Geographic patterns of migratory routes

Figure 5.1 shows the wildebeest migration as it used to be in the early post colonial times. Wildebeest migrated in all but the Serengeti-Mara ecosystem, from within protected areas in the dry season to dispersal areas outside in the wet season. Figure 5.2 shows the status of these migratory routes in the 2000. The figure reveals that migration has discontinued altogether, and that it was reduced along a number of other routes.

It is noteworthy that this happened where wildebeest migrated outside protected areas. No discontinuation or reduction of migration is reported from the Serengeti-Mara ecosystem where wildebeest migrates almost entirely within protected areas. The Figure 5.2 further reveals that discontinued or reduced migration routes overlapped with agricultural and settlement expansion in the Mara, Loita-Siana and Tarangire ecosystems and fences and settlements in the Athi-Kaputiei Plains. The figure does not give evidence for agriculture, settlements or fences as a cause of change in some migratory routes in the Amboseli ecosystem.

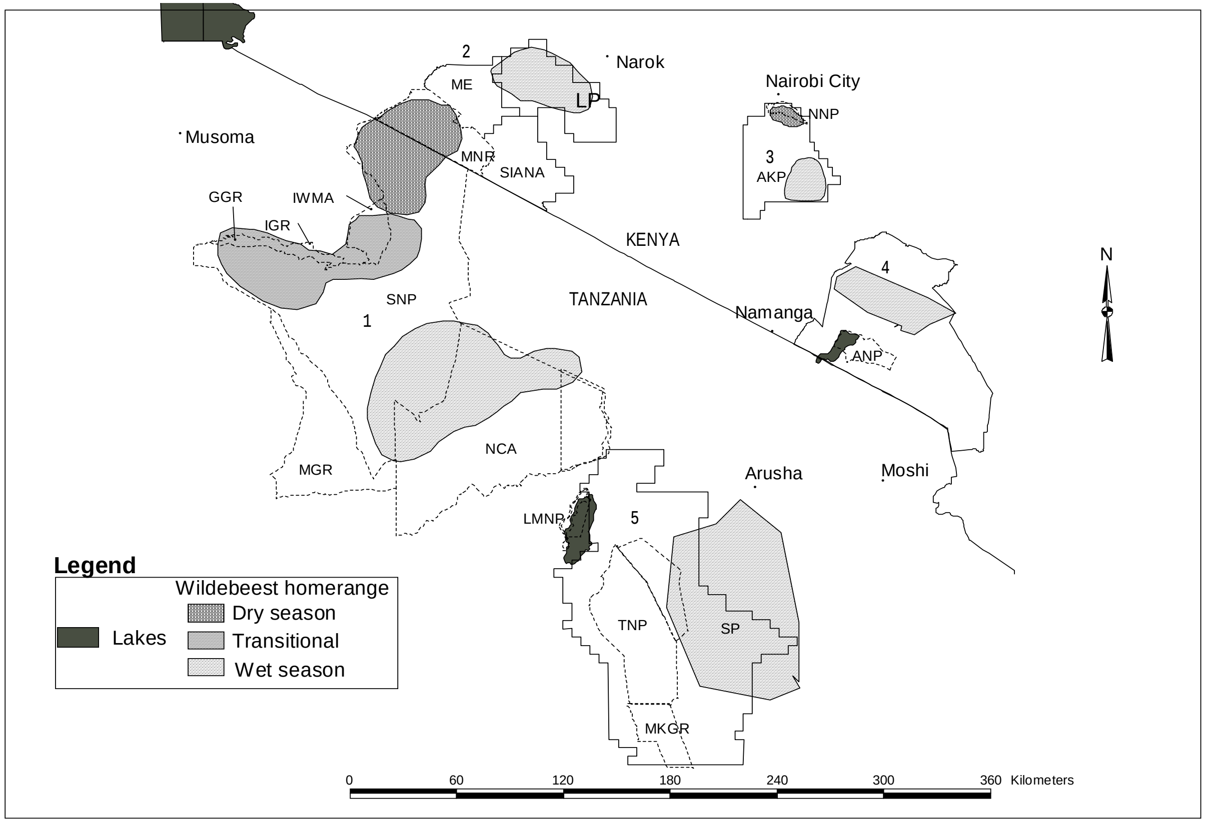

Figure 5.1 Map showing the five ecosystems in East Africa with distinct migratory wildebeest population; 1= Serengeti-Mara, 2=Mara-Loita-Siana, 3= Athi-Kaputiei, 4= Amboseli and 5= Tarangire.

Note: SNP=Serengeti National Park, NCA=Ngorongoro Conservation Area, MGR=Maswa Game Reserve, GGR & IGR=Grumeti and Ikorongo Game Reserves, IWMA=Ikoma Wildlife Management Area, MNR=Masai Mara National Reserve, LP=Loita Plains, NNP= Nairobi National Park, AKP= Athi-Kapiti Plains, ANP= Amboseli National Park, LMNP= Lake Manyara National Park, TNP= Tarangire National Park, SP= Simanjiro Plains and MKGR= Mkungunero Game Reserve

Figure 5.2 Distribution of agriculture in the Mara-Loita-Siana and the great Serengeti-Mara ecosystems (a), Tarangire (c), Amboseli (d) and fences distribution in the Athi-Kaputiei ecosystem (b) and the status of wildebeest migratory routes in the post 2000 period.

5.3.2 Trends in wildebeest population

Figure 5.3 shows the temporal trend of the wildebeest population in the Serengeti-Mara ecosystem over a span of 55+ years. The population grew steadily since the disappearance of the rinderpest in 1963 (Sinclair *et al.* 1985), until the late 1970s when it started to oscillate (decrease and increase) with one noticeable decline in the early 1990s, when a severe drought reduced the population from around 1.2 million to less than 900,000 animals (Mduma *et al.* 1999). The population has since then continued to recover and stabilized at around 1.3 million animals (Thirgood *et al.* 2004).

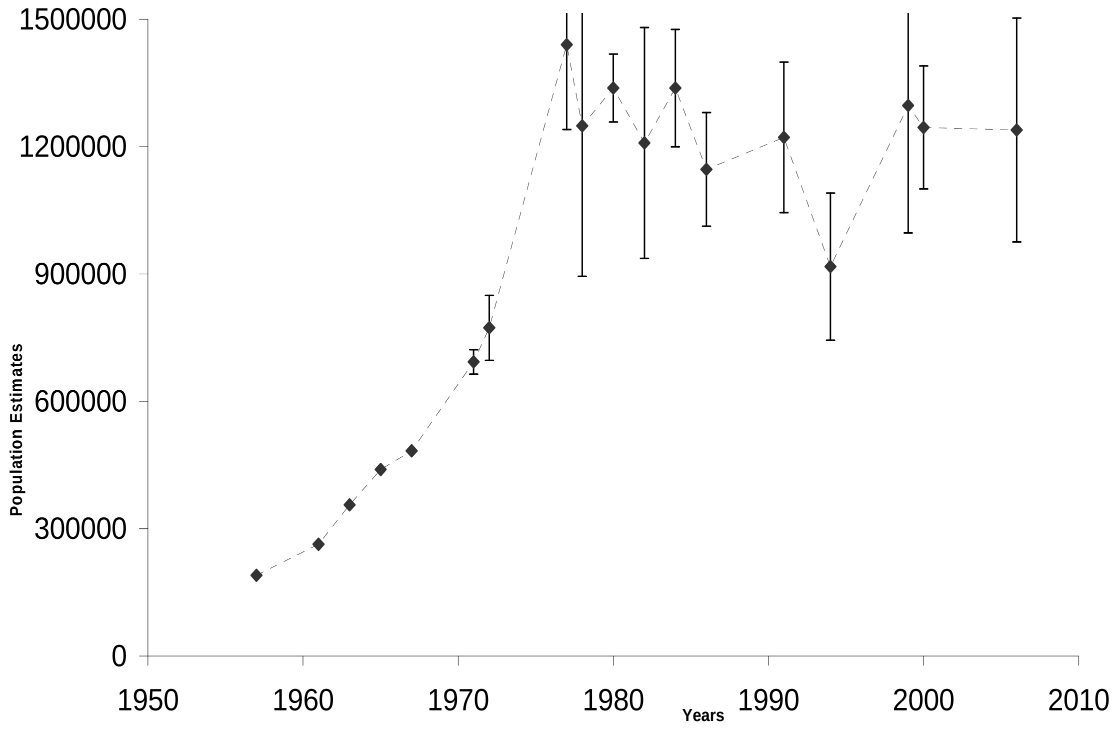

Figure 5.3 Long-term trend population estimates of wildebeest in the Serengeti-Mara ecosystem (Data from FZS & TWCM in Serengeti National Park).

Figure 5.4.a shows the migratory wildebeest population in the Amboseli ecosystem. The population numbered around 25,000 in the late 1970s, fluctuated between 15,000 and 20,000 individuals till early 1980s and fell below 15,000 thereafter and later increased to around 35,000 by early 1990s. The population declined to less than 5,000 in 2010 following a severe drought in the preceding 2 years. Table 5.2 shows the wildebeest population trends in the four ecosystems (Amboseli, Mara-Loita-Siana, Athi-Kaputiei and Tarangire) and indicate that the population decline in Amboseli was significant ($F = 11.81$; $p = 0.0056$).

The Mara-Loita-Siana population has been declining steadily between the late 1970s and early 1990s falling from around 150,000 to 40,000 (Fig. 5.4.b). Table 5.2 also support these observations by showing that the decline was highly significant ($F = 128.9$; $p < 0.0001$).

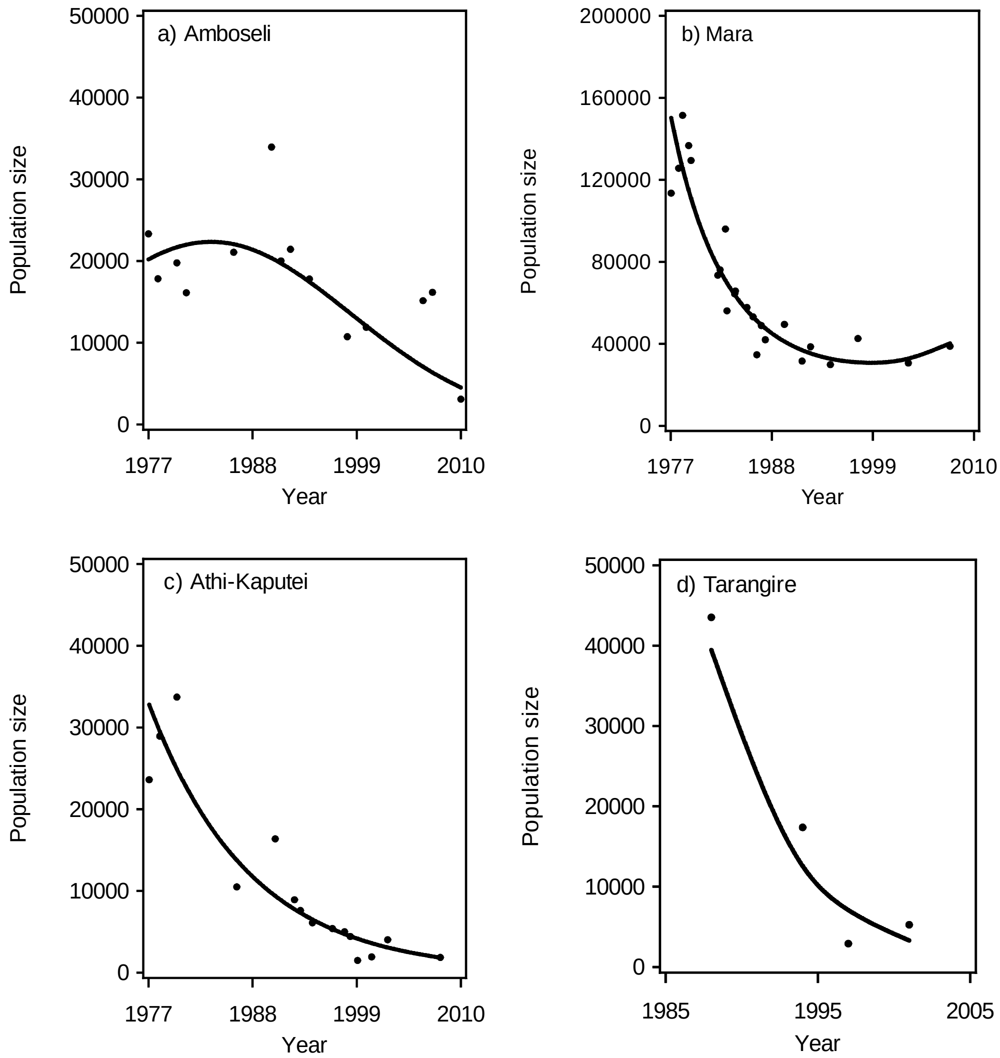

Figure 5.4 Time series population trends of migratory wildebeest in Amboseli (a), Mara-Loita-Siana (b), Athi-Kaputiei (c) and Tarangire (d) ecosystems.

The Athi-Kaputiei wildebeest population has followed a similar trend, although the initial population was smaller than that of the Mara-Loita-Siana. The decline of this population has been more dramatic in recent decades, leading to a near collapse of the population (Fig.5.4.c). The wildebeest numbers declined from around 30,000 in 1977 to less than 10,000 by mid 1980s. The population increased again to about 10,000 by early 1990s and later declined to under 2000 individuals. Table 5.2 shows that the decline was highly significant (F= 84.6; p <0.0001).

The Tarangire wildebeest population has experienced a precipitous fall and has shown no signs of recovery since the late 1980s, with the numbers declining from around 45,000 to a mere 5,000 individuals within a period of less than 15 years (Fig. 5.4.d). This extreme decline was however not statistically significant (F= 11.71; p = 0.076) due to large variances of the population estimates from the aerial surveys and small sample size (Table 5.2).

Table 5.2 Regression statistics describing the relationships between the natural logarithm of population size and year of survey

Region	Number of surveys	Intercept	Linear Slope	Quadratic slope	F	P>F
Amboseli	14	10.01	0.0063	-0.0023	11.81	0.0056
Mara-Loita-Siana	23	10.96	-0.093	0.00342	128.9	<0.0001
Athi-Kaputiei	15	9.63	-0.099	-	84.65	<0.0001
Tarangire	4	11.16	-0.191	-	11.71	0.076

5.4 Discussion

This study revealed that wildebeest migratory routes/corridors and populations have declined significantly, in three out of five ecosystems where wildebeest migrate in East Africa, comprising the Mara-Loita-Siana, Athi-Kaputiei, and Amboseli ecosystems. The Tarangire population has also declined dramatically even though the statistical test failed to reach significance due to low test power arising from high variances of population estimates and a small sample size.

Further, the analyses revealed that two important processes, i.e. agricultural expansion and fencing (see also Table 5.3) exclude wildebeest from their grazing resources. In two out of the four ecosystems (i.e. Mara-Loita-Siana and Tarangire), where migratory wildebeest are declining, agricultural encroachment ranks as an important factor excluding wildebeest from forage. Settlements are also an important interference in the Mara (Lamprey & Reid 2004) and Tarangire Msoffe *et al.* (in press) as it blocks the migratory routes. Agriculture also encroached in the Amboseli ecosystem, but does not apparently overlap with wildebeest migration routes. It is therefore likely a less important driver of the dynamics of wildebeest in the Amboseli. Fencing and expansion of settlements are the primary factors excluding wildebeest from grazing resources in the Athi-Kaputiei ecosystem (Table 5.3)

Table 5.3 Summary of the processes associated with the declining migratory wildebeest populations and patterns in the East African rangelands

Processes	**‡Serengeti-Mara**	**Mara-Loita-Siana**	**Athi-Kaputie**	**Amboseli**	**Tarangire**
Direct interferences/causes					
Agricultural encroachment	-	+++	+	+++	+++
Fencing	-	-	+++	++	-
Settlements	+	++	+++	++	++
Urbanization	-	+	+++	-	-
Roads & Infrastructural developments	+	++	+++	+	++
Poaching	+	+	++	+	++
Drivers					
Human population increase	+	++	+++	++	+++
Land tenure change	+	++	+++	+++	+
Settlements policies	++				+++

‡+++ High importance; ++ Important; + less importance; - not important

Source: Interviews with resident researchers; Reid *et al.* (2008); Msoffe, *et al.* (in press).

Apart from agriculture and fences, other factors suggested to adversely affect the migratory wildebeest populations by limiting their access to food and water include human population growth, land subdivision and privatization of land tenure, sedenterization of formerly semi-nomadic pastoralists and intensification of land use (Reid *et al.* 2008,Ogutu *et al.* 2009). Human population growth and expansion of settlements are most pronounced in the Athi-Kaputiei ecosystem because of expansion of the Nairobi Metropolis, rapid urbanization in the Athi-Kaputiei Plains, and lower land prices and land rents compared to Nairobi City. Development of new industries, businesses and infrastructure attracts people from Nairobi City and rural areas searching for employment and residential areas (Reid *et al.* 2008). Fencing of individual land parcels, water points, commercial flower farms, tree farms and settlements have severely impeded wildebeest movements between Nairobi National Park and the Athi-Kaputiei Plains (Reid *et al.* 2008, Ogutu, *et al.* in prep). Agriculture was previously practiced mainly by outsiders and not the local Maasai pastoralists but the Maasai recently started cultivating next to their pastoral settlements. The adoption of small-scale subsistence agriculture is taking place all over Maasailand, and is potentially a threat for wildebeest migrating outside protected areas, and urgently deserves further study to documents its impacts. Previous studies have also indicated agriculture, particularly the large-scale cultivation as the major cause of habitat loss for the migratory wildebeest. For example (Serneels & Lambin 2001b) showed that about 50,000 ha of natural vegetation were converted to wheat farms between 1975 and 1995 in the Loita plains of the Mara-Siana-Loita ecosystem. In the Tarangire ecosystem, about 71,000 ha of land were converted from rangelands to farms (mostly large-scale seed beans and maize cultivation) between 1984 and 2000 (Chapter 2, Msoffe *et al.* in press). Such habitat conversions removed a large portion of forage and dispersal area used by migratory wildebeests.

Land tenure change from group ranches to private ownership has been suggested as another important factor besides agricultural encroachment (Thompson & Homewood 2002). The land sub-divisions associated with fencing in Athi-Kaputiei and Amboseli ecosystems are causing habitat fragmentation and directly interfering with the migratory wildebeest (Reid *et al.* 2008,Western *et al.* 2009a). Villagization of settlement policies in Tanzania has been identified as a key driver causing blockage of migratory routes due to conversions to agriculture in the Tarangire ecosystem

(Msoffe, *et al*., in press). It might also have an impact on the Serengeti-Mara ecosystem population, particularly in Western Serengeti, in the Ikoma Wildlife Management Area (Thirgood *et al.* 2004), where poaching of migratory herds is also associated with increasing human population size (Mduma *et al.* 1999).

Agricultural encroachment and fencing are two important processes causing the decline of the migrant wildebeest populations in the four ecosystems. These processes constrain wildebeest access to grazing resources, water and calving areas. In Kitengela, a major part of the Athi-Kaputiei plains, adjoining Nairobi National Park, fenced land parcels have spread throughout the range of wildlife and movements of people, livestock, dogs and vehicles harass wildlife (Reid *et al.* 2008). The restrictions on mobility have negative consequences for migratory wildebeest especially during droughts when heavy mortality results wildebeest access to water and food is blocked (Williamson & Williamson 1985,Tambling & Du Toit 2005).

Wildebeest populations differ from many other wild herbivores in that they have a strong preference for land with a high potential for agriculture (Norton-Griffiths 1996) and therefore competes with both agriculture and livestock grazing in pastoral rangelands. This raises doubts over the effectiveness of community based approaches popularly promoted to conserve wildlife in rangelands for conserving migratory wildebeest populations (Western 1982,Kideghesho 2002,Thirgood *et al.* 2004).

There are several reasons to suspect that community based approaches will be less efficient in reducing the loss of wildebeest habitats in pastoral lands. The first is that wildebeest competes for land which is suitable for agriculture, which tends to generate higher economic returns than livestock or conservation (Norton-Griffiths *et al.* 2008). Thus rational land users will often opt for cultivation rather than conservation, driving the encroachment of agriculture and the associated declines by wildebeest populations. The second is that, once land has been cultivated it is extremely difficult to restore it to its former rangeland status because returns from agriculture are higher than those from other land uses.

What can be done to revert these trends and allow wildebeest access to their former habitats? Financial subventions are needed to generate incentives sufficiently competitive with agriculture to motivate land owners to restore their lands to accommodate the migration of wildebeest. It is clear that the budgets for such subventions need to be provided from outside pastoral systems. Where it is deemed to be in the national interest to conserve wildebeest populations, for example while maintaining a viable tourism industry, governments might do well to consider purchasing easements on occupied land used by wildebeest as dispersal areas.

In conclusion, the populations of migratory wildebeest in four out of five key ecosystems in East Africa are under threat and two populations might be on their way to collapse if the trend is left to continue. Agricultural encroachment and fencing are the major threats responsible for this as these reduce wildebeest access to grazing resources, water and calving areas. This study therefore suggests that governments need to take the lead in conserving the remaining key wildebeest habitats to ensure continue access to grazing resources in these rangelands. Such measures might include more efforts to establish wildlife conservancies and/or management areas by key stakeholders to include migratory routes/corridors and calving grounds for the species. Where migration occur across trans-boundaries (such as the Serengeti-Mara ecosystem) it is important to harmonise wildlife policies to ensure the long-term survival of migratory species and the sustainability of the rangelands in such ecosystems.

6 Chapter Six: Thesis Summary and Discussion

6.1 Introduction

This thesis has tackled the underlying drivers, forces and proximate causes associated with changes observed in the Maasai-rangelands and their impacts and consequences, proposed a monitoring strategy locally and also extended the study further to a meta-population of a migratory species for generic solutions across the rangelands. This research project was part of a regional study undertaken by ILRI, focusing at broad patterns of trends in wildlife and land use across savanna ecosystems of the Maasai land in Kenya and Tanzania. The study aimed at providing recommendations for land-use options, policy changes and improvements geared at reducing the adverse impacts of land-use changes on biodiversity while sustaining the agro-pastoral livelihoods and also at reducing the human wildlife conflicts.

This research focused at the landscape and local scale level, looking more on the processes related to understanding the drivers and dynamics of land-cover conversions and land-use change, their consequences and impacts on the large migratory herbivores and agro-pastoralists in the Maasai-Steppe ecosystem of Northern Tanzania. The work extended to other four key ecosystems in the East African savannas, with migratory species, in order to assess the drivers and impacts of land use change on the population and migratory patterns and to suggest possible interventions for the conservation of migratory species covering both protected and unprotected lands. The following section summarizes the main results from the thesis, followed by a discussion of relevance of these findings.

6.2 Summary of key findings from the data chapters

In Chapter 2 an integrative approach was used to unfold and synthesize the drivers and causes of land-use change emanating from historical, political and livelihood needs. Results indicated that agricultural expansion and human population increase were in parallel with declining routes and corridors for key migratory wildlife species. Recurrent droughts and diseases also contributed to the declining livestock economy from livestock losses and the unpredictable and erratic rainfall limited their recovery. The results of this study reinforce findings of other studies conducted in East Africa indicating that wildlife habitats inside and outside protected areas are at a high risk of becoming ecological islands able to support a fraction of the previous wildlife populations (Nelson 2007,TNRF 2008,Western *et al.* 2009b). The land transformations currently underway in Tarangire ecosystem are similar to those observed in Kenya over the last 20 years and portend grave consequences for the future of migratory wildlife populations and pastoral livelihoods.

Furthermore, Chapter 2 shows that changes in land tenure policies and human population growth are the main drivers of the land use change, which triggered the degradation of the pastoral livelihoods and the wildlife resources. Changes from customary and communally open rangelands to exclusive use of gazetted protected areas by wildlife only (particularly the dry season grazing and watering areas) excluded pastoralists and separated them from wildlife conservation. The villagisation policy further exacerbated the problems because it forced the agro-pastoral communities to settle (sedentarization) with their livestock, decreasing their freedom to move within the rangelands and to cultivate more for their livelihoods.

In Chapter 3 it is shown that there is a strong overlap between lands suitable for agriculture and wildlife, particularly key migratory species, such as wildebeest. Results from Chapter 3 showed that people tend to prefer the same habitats as other large mammals, landscapes with reliable water sources and moderate rainfall. Expanding cultivation towards protected areas severely restricted wildlife movements to dispersal areas outside parks by blocking their migratory routes. Further, the global model used for the prediction of probability of land-conversions to agriculture suggested future expansions will be constrained by values of the biophysical variables, such as rainfall and settlements. The speed of rangeland conversions to farming presents a major threat to wildlife conservation and disrupts the ecosystems viability in supporting its rich biodiversity and the agro-pastoral livelihood. The diminishing range size implies that small stochastic events such as droughts could affect larger proportions of wildlife and livestock populations, especially for mammals that directly threaten human lives and/or compete with humans for resources and if they are restricted inside artificial boundaries (Thuiller *et al.* 2006), as in this case due to blockage of their migratory routes.

Chapter 4 demonstrated that protected areas alone are not sufficient to conserve wildlife populations, particularly migratory species. The importance of involving local communities in monitoring programs across landscapes that incorporate communal lands as well as protected areas is inevitable under the current observed land-use change and wildlife trends. In this survey, the five most abundant species were wildebeest, gazelle, zebra, cattle and people. This implies that in order for them to continue to co-exist in the rangelands there has to be harmonization of the different land-uses for the wildlife, livestock and people who are also cropping. It follows that the development and evaluation of ground-based survey methods that require inexpensive low technology equipment and allow the widest participation of community stakeholders could enhance conservation goals on communal lands.

Results from Chapter 4 showed that, estimated densities from the ground survey provided more reliable baseline data to assess the population status of key species in the ecosystem than estimates from aerial surveys. In addition, because DISTANCE sampling on walked transects can focus on small discreet areas, it is possible to conduct censuses in hunting blocks and communal areas to provide valuable information to guide management activities (Waltert *et al.* 2006). There is also increased interest and debate concerning the decentralization of management of wildlife resources (Gaidet-Drapier *et al.* 2006,Nelson 2007), which in this case will empower communities in managing wildlife resources on their land and increase a sense of ownership.

Chapter 5, investigated the status of migratory routes/corridors, dispersal ranges and population trend of a key migratory species in five ecosystems within East African rangelands. The motive behind this was to come up with a synthesis of one key species meta-population analysis in this study and use it as a stepping stone to come up with recommendations on intervention measures in the conservation of migratory species which covers protected and the unprotected lands. Significant findings showed a highly threatened population in four of the five studied ecosystems. Agricultural encroachment and fencing excludes the species from accessing grazing resources and also fragmentation through land sub-divisions and permanent settlements continue to take up their key habitats.

As human population grows, with East African projections expected to double by 2050 from the current 327 million to 711 million (ESA-UN 2009), settlements and agriculture expand through the most desirable areas, and then push into increasing marginal territories. Such increases will have huge impacts and pressure on the rangelands and natural resources. In East Africa, this means settlements and farms moving into semi-arid and arid areas that are not only wildlife rich, but have been the traditional home of pastoralists. Conflict arises as people clear land for cultivation, destroying wildlife habitats and livestock grazing areas.

Results from Chapter 5 showed that the cause of the drastic wildlife population declines has undoubtedly been the twin threats of habitat loss (mainly from agricultural encroachment and fencing) and poaching in the dispersal areas and calving grounds for the migratory species (Thirgood *et al.* 2004). While the relative impact of each threat is unclear, it is likely that these two processes are becoming increasingly linked, and the rate of decline appears to be accelerating These animal losses are unsustainable; if the decline is not reversed or slowed, the main migration will soon cease to exist.

6.3 Implications of the results in the broader context

The results from this thesis have major implications, not only in the rangelands of East Africa, but worldwide, where major land-use changes are occurring at unprecedented speed, causing disappearance of wildlife species and biodiversity loss. However, according to the United Nations human population growth predictions, most future global population growth will happen in Africa with the greatest pressure expected on high biodiversity areas (ESA-UN 2009). Land use is expanding and intensifying in unprotected lands surrounding many of the world's protected areas. In the western United States 11 out of 13 national parks have lost large mammal species, ranging between 5-21% of the original species since their establishment because many of these areas have been altered and have lost their original critical ecological functions required by the species (Parks & Harcourt 2002). In the tropics, road construction, conversion for agriculture and demand for natural resources have lead to clearing of primary forests around reserves and increased hunting of native species

(Hansen & DeFries 2007). In China, agricultural and urban land uses have continued to push into unprotected wildlands around protected areas (Vina *et al.* 2007). In the case of China, although the concern was on a particular keystone species, i.e. the giant panda in Wolong Nature Reserve, Sichuan, the consequences might also have impact on other species. This thesis adds to the body of literature describing the drivers and impacts of land use change in relation to the conservation of large migratory ungulates utilizing both protected and unprotected lands.

The role of the savanna rangelands in supporting the economy of countries cannot be underestimated, because tourism has been the biggest foreign exchange earner from millions of tourists in both Kenya and Tanzania over the last decade, supporting thousands of livelihoods (FAO 2009). In Tanzania, wildlife is the key attraction for a tourism industry that drew over 700,000 visitors in 2007, and over one billion U.S. dollars (TNRF 2008). Tourism in turn has been vital to economic recovery and growth of the past two decades. Visitors to Kenya totaled over two million in 2007, and accounted for about 12 percent of the Gross Domestic Product (Reid *et al.* 2008). In Tanzania, it has been shown that wildlife tourism primarily from game viewing and tourist hunting are the most lucrative legal forms of wildlife use which can fully contribute to the national economy (Leader-Williams 2000). Local communities living among wildlife outside protected areas in Tanzania can also benefit equally from wildlife tourism because the fees paid in any hunting operations is equal in both areas (i.e. protected and unprotected land). Further benefits through employment in the tourist sectors (for example, hotels and lodges, camping), agricultural sector from food supply and selling of artifacts are just a few to mention. It is however, ironic, that government policy does not support pastoralism equally as it does support farmers who are purely cultivators, and yet it is clear that, pastoralism is the most compatible and viable land-use with wildlife conservation in these ecosystems. So pastoralists too are beginning to farm even in those semi-arid areas where pastoralism with mobility remains the most suitable land-use option (Reid *et al.* 2003). It is also apparent that if significant financial benefits from wildlife are not provided to these rural people living among wildlife areas, illegal use of wildlife will continue despite conservation efforts by the wildlife institutions (Leader-Williams 2000).

On the other hand, economic analysis shows that in the current policy environment, farming reaps higher returns per hectare than wildlife even in areas that see the highest numbers of wildlife tourists, for example, in the Mara ecosystem in Kenya (Norton-Griffiths *et al.* 2008). Unless local communities see that wildlife benefits them at least as much as farming, they will continue to farm and/or lease or sell their land to others who will farm, either for subsistence or commerce. This is a challenge to governments, that without policy change, economics will continue to drive out wildlife. Chapter 5 suggests the need to bring in incentive based conservation initiatives and measures, which can be competitive and motivate land-owners (private or community) to restore lands in order to accommodate wildlife. As (Hutton & Leader-Williams 2003) puts it, "that sustainable use include direct use as an imperative or choice, the ideal of keeping any use within biologically sustainable limits, and use as possible conservation strategy that can create positive incentives, which are key where land could otherwise be converted to biodiversity-unfriendly practices".

In Tanzania, Wildlife Management Areas (WMAs) were mandated by official policies, including the Poverty Reduction Strategy which recognized that the best-suited land use option to generate funds for WMAs in many parts of Tanzania would be tourist hunting (Baldus & Caudwell 2004). However, local communities are increasingly frustrated that promised benefits from tourist hunting, and the promised reform of wildlife policy to established WMAs, have not been forthcoming as has been popularized in the last 15 years. Such frustration may in turn encourage local communities to revert to poaching or habitat conversion, unless reforms are forthcoming (Leader-Williams *et al.* 2009). However, before WMAs can operate effectively and bear fruits, there is a need for radical changes and reforms in policies and legislations particularly in the wildlife and land sectors. It is inevitable that such changes and reforms are considered in the face of the ever changing environment and the human population growth particularly in high biodiversity areas. For this to happen, economic and viable forms of wildlife use should be identified relevant to the area, and which can also be practiced in a sustainable manner, in the face of competing land uses (Leader-Williams 2000).

Governments can provide incentive schemes to local communities in traditional grazing lands in order to keep areas for grazing purposes only (both livestock and wildlife) and hence another way of opening up migratory corridors/routes and calving areas for wildlife. In addition, community-based wildlife tourism projects such as visiting traditional '*Maasai-bomas'*, selling of artifact and camping in community lands can provide more tangible benefits at the household levels and because they are more likely to be incentive driven in unprotected lands they can be a better way forward for sustainable land use practices. Government support is needed to spearhead the formation of wildlife conservancies and management areas by consolidating different land-uses in a more holistic management approach. This should be envisaged even across trans-boundaries, where wildlife requires habitat that covers beyond national boundaries, such as for those migratory species covered in this study. The delaying in the implementation of operational WMAs which currently need reforms in order to give the local community more power and autonomy in managing the resource is going to have much more negative consequences on biodiversity conservation.

6.4 Suggestion for further research

The following is a summary of areas where further research was considered necessary from this thesis.

Analyses of the ecosystem services in the rangelands would be an important aspect in order to quantify and qualify the values of the different functions, goods and services provided by these ecosystems. This should be possible given the premise of availability of high temporal and high resolution remote sensing data from current censors, such as the Moderate-Resolution Imaging Spectroradiometer (MODIS) and the Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER). In theory, if we can help communities and institutions to recognize the value of nature; it should greatly increase investments in conservation, while at the same time fostering human well being (Daily *et al.* 2009). The Millenium Ecosystem Assessment advanced a powerful vision for the future in which people and institutions appreciate natural systems as vital assets, recognize the central roles these assets play in supporting human well-being, incorporating the tangible and intangible values (Daily *et al.* 2009). It would be beneficial to quantify the value of the different habitats for carbon sequestration, water retention, medicinal and cultural values of certain species and their economic values as well. Similarly the costs and benefits of the different land-uses such as cultivation, livestock grazing versus wildlife conservation such as eco-tourism and hunting, in the long-term, in order to understand their trade-offs in the ecosystem. Equal emphasis needs to be placed on including effective regimes that also encompass private and communal ownership through incentive-based approaches (Leader-Williams 2002).

Further research should also focus on the design of wildlife management areas/community conservancies (WMAs) in these ecosystems. Their design should be connected to the above in that such areas would clearly represent areas of high importance in terms of wildlife conservation and other potential land-uses but also mitigating conflicting land uses. For the case of migratory wildlife species, such as wildebeest they should as much as possible coincide with their corridors/routes and calving areas. This will provide the basis for decision making and in coming up with participatory land use plans in a more empirical manner. The delay in the implementation of WMAs policies in most of the Tanzania's wildlife rich areas has been tied to lack of funds in carrying out land use planning which is central and key requirement among others for the establishment of operational WMAs.

Another area of particular concern is on wildlife hunting. Research particularly focusing on the impact of hunting (both tourist and resident hunting) on migratory species and other species would be necessary in these ecosystems. As shown above, tourist hunting is one of the most lucrative business that could generate revenue from wildlife directly with minimal impact on the environment (Leader-Williams 2000), as compared to other forms of land-uses in most of the Tanzanian rangelands. In parallel with would be a monitoring system, as a priority and should be put in place accordingly for setting the hunting quotas to ensure sustainable utilization of the different species, as well as understanding the effects of illegal hunting (poaching) on different species, including the migratory populations.

6.5 Conclusion

This thesis has tackled the main underlying drivers and causes of land-use changes, their consequences and impacts on biodiversity conservation and pastoralism, which have co-existed for centuries in the Maasai rangelands. Historically a number of conservation and development policies have affected the way that conservation has been conducted in Tanzania. Development policies including re-settlement and villagisation, have broken down traditional structures and reduced communal ownership of land and natural resources. Conservation policies and legislation have further precluded the interests of local communities from legal forms of wildlife utilization. As a result of these policies, agro-pastoral communities living around protected areas have lost the long-traditional culture of cohabiting with and tolerance towards wildlife on their land. They now see wildlife as competing with them on their land and the only way to deter them is to convert these rangelands into croplands. Proposed national policies such as the new category of wildlife conservation outside protected areas (WMAs) will need to be implemented effectively to ensure that local communities are involved in, and receive more direct and tangible benefits from, the management of wildlife resources in areas where they live.

In the current policy environment, agriculture is often the most lucrative land use in the short term. But in the semi-arid rangelands of East Africa, it is often unsustainable and ends up degrading soils to the point that neither farming nor grazing can continue. Sustainable livelihoods need urgent support in the study area. Increasingly rapid conversion of rangeland into farmland is leading to a plunge in wildlife populations, despite their national economic importance and the decline of pastoralism as a livelihood, despite its sustainability in semi-arid regions. Reversing unsustainable cultivation and declining wildlife trends would require rigorous measures by the government, NGOs and support from local communities.

The conservation of these areas hence, lies on the hands of the local, regional and the international community. Where it is deemed to be in the national interest to conserve wildlife populations, for example while maintaining a viable tourism industry, governments might consider buying out farmers who have occupied wildlife dispersal areas. This should happen now, as it would be much more complex if not impossible in the future to restore these rangelands and wildlife habitats given the scale and rates at which land use change and conversions are currently underway.

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8 Appendices

Chapter Two Appendices

2A- Temporal behaviour of NDVI for images selection

2B- Details of images classification

APPENDIX 2A

Temporal behaviour of the NDVI in the Tarangire ecosystem for images selection

The Normalized Difference Vegetation Index (NDVI) provides a measure of the amount and vigor of vegetation at the land surface. The magnitude of NDVI is related to the level of photosynthetic activity in the observed vegetation. In general, higher values of NDVI indicate greater vigor and amounts of vegetation. NDVI is derived from data collected by National Oceanic and Atmospheric Administration (NOAA) satellites, and processed by the Global Inventory Monitoring and Modeling Studies group (GIMMS) at the National Aeronautical and Space Administration (NASA). The data was downloaded from the Africa Data Dissemination Service (http://earlywarning.usgs.gov/adds/) and the monthly NDVI values for the study area were extracted and plotted against time.

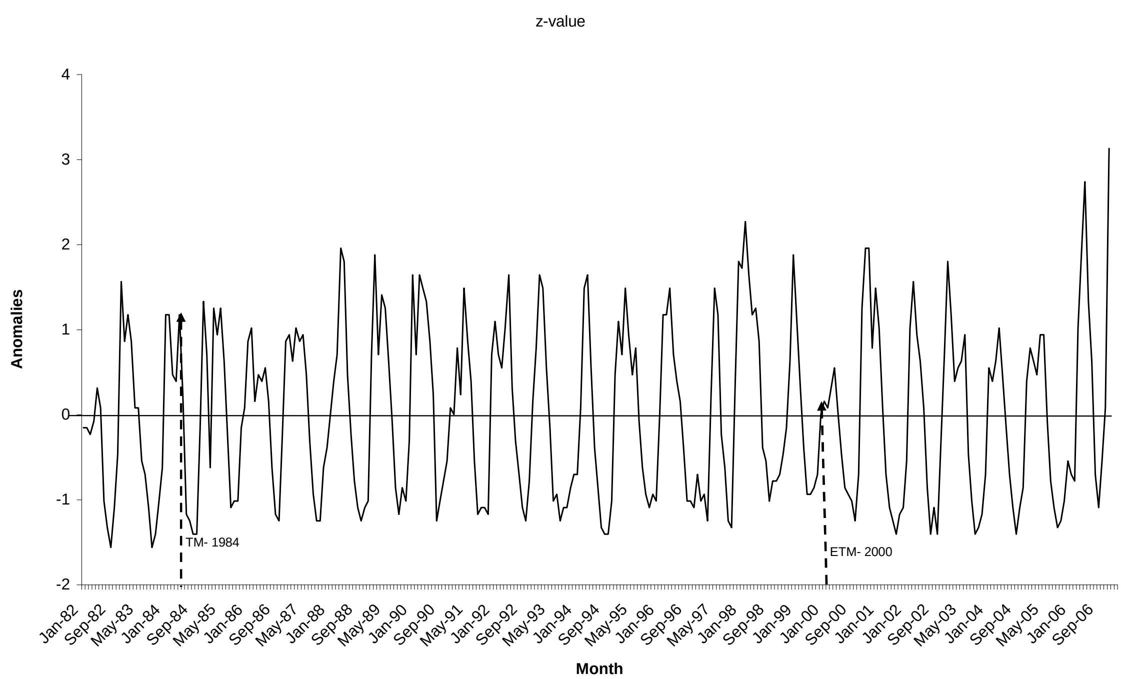

Figure 2A.1Temporal behavior of NDVI in the study area between 1982 and 2006. Dashed vertical arrows indicate the probable time for the acquisition of the two images

APPENDIX 2B

Details of Images classification

(1) Two mosaics were created from images with no clouds, after masking areas with clouds. The mosaics were; 1. P168r62_84 and P168r63_84 2. P168r62_00 and P168r63_00.

(2) The mosaics were then clipped by study area boundary

(3) The using Erdas imagine the two mosaics resulting from (5) were subjected into unsupervised classification and 100 classes were created.

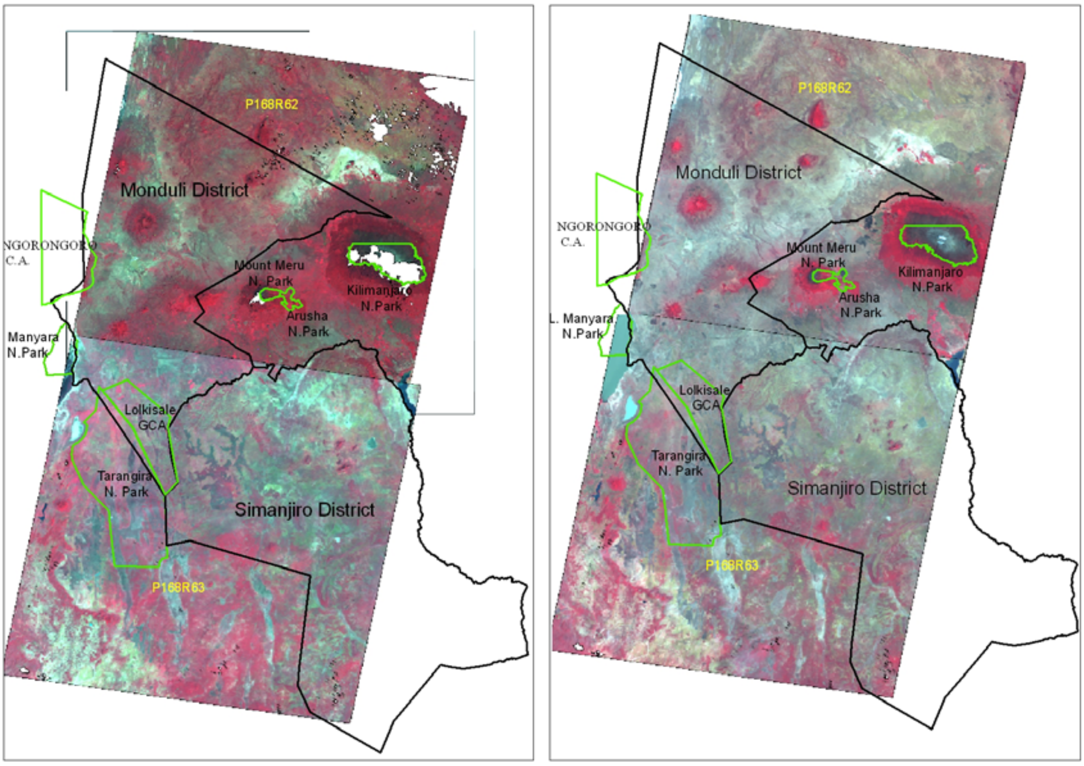

Figure 2B.1 Satellite images overlaid with the study area. Deep red coloration shows areas of dense vegetation and light bluish are more opened vegetation

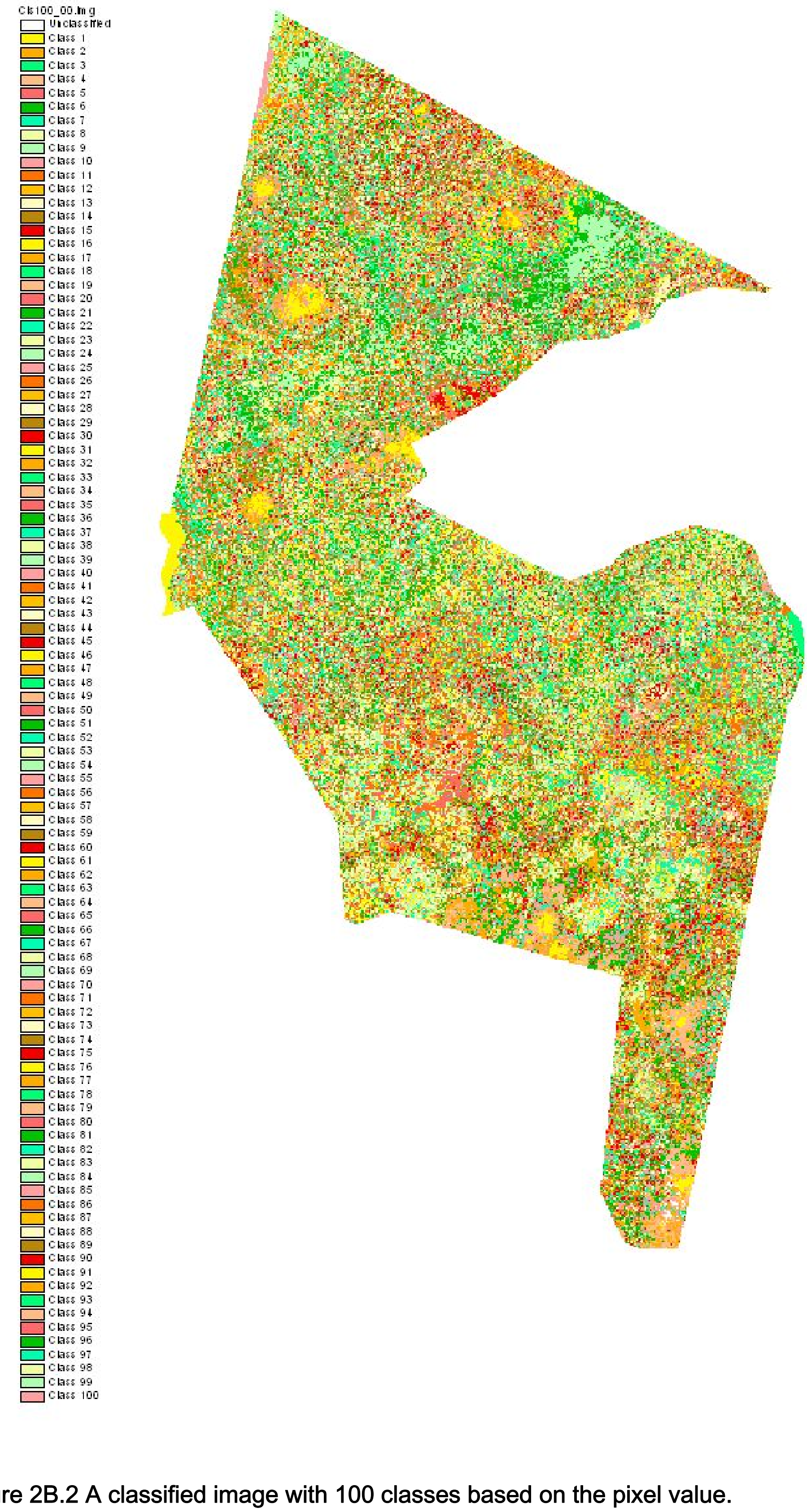

Figure 2B.2 A classified image with 100 classes based on the pixel value.

(4) Each of the classes was assigned a landcover class name mostly based on Africover classification classes but there were difficulties because Africover was interpreted visually from hard copy prints and there was a lot of generalization hence in our case we are working at pixels levels. In this case so many classes could fall into one Africover class.

- A method of getting 3 classes with majority pixel was adapted within Africover polygon and the 3 classes were assigned to the class from the Africover polygon.

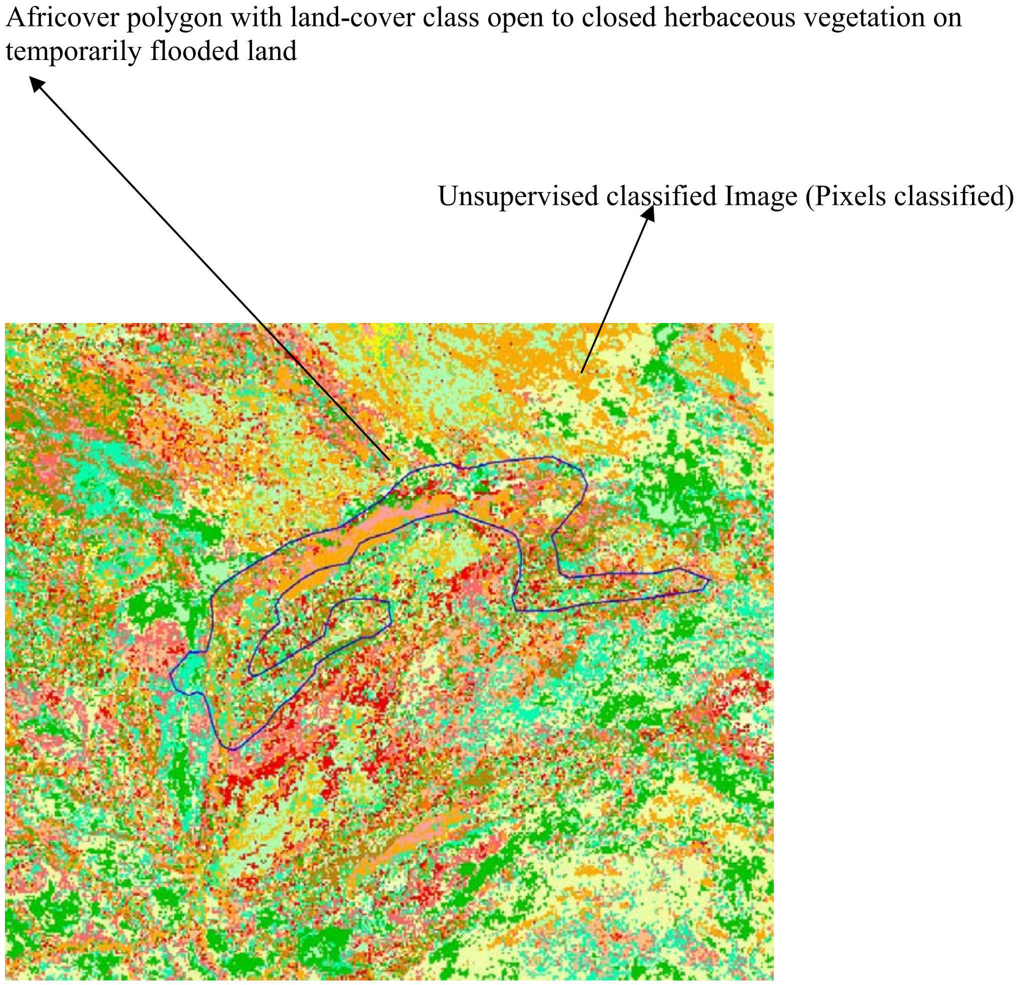

Figure 2B.3 The map shows area outline in blue classified as open to closed herbaceous vegetation on temporarily flooded land, while the unsupervised classification assigns four classes based on pixel value. A combination of these pixel values was used to classify the image into broad land cover classes.

Zone	Open to closed herbaceous vegetation on temporarily flooded land
Class 17	503
Class 97	490
Class 32	412
Class 37	391

* Include %

Classes 17,97,32,37 are within the blue Africover polygon and the majority in that order. Each of the classes above was assigned Africover land-cover class name ***Open to closed herbaceous vegetation on temporally flooded Land***

(5). Small and homogenous polygons in colour from Africover database were considered to avoid a big variety in pixels classes within the polygon. The above procedure was repeated several times until the 100 pixel classes were assigned a Land-cover class from Africover Land-cover data. We could not exactly apply the same method to classify the 1984 mosaic, since no Land-cover data or maps were done in the same year or there about which are available to guide in Land-cover classes assignment. We therefore decided to use 2000 image characteristics to classify the 1984. Areas in both images with same characteristics were identified and pixels classes within the area in the 1984 image were assigned land-cover class to that 2000 image. This procedure was repeated until all 100 classes were assigned land-cover classes for both images.

To map agricultural land use change between 1984 and 2000, the following three classes for land use were identified: (1) areas with agriculture in 2000 but not in 1984, (2) areas with agriculture in both 1984 and 2000 and (3) areas with agriculture in 1984 but not in 2000.

3.3 Field verification

For each of the three scenarios above, several sample points were picked based on their proximity to roads and were given unique identifiers. The total number of points picked for sampling was 214 out of which 177 were sampled and 82 out of 177 were found to correspond with initial interpretation.

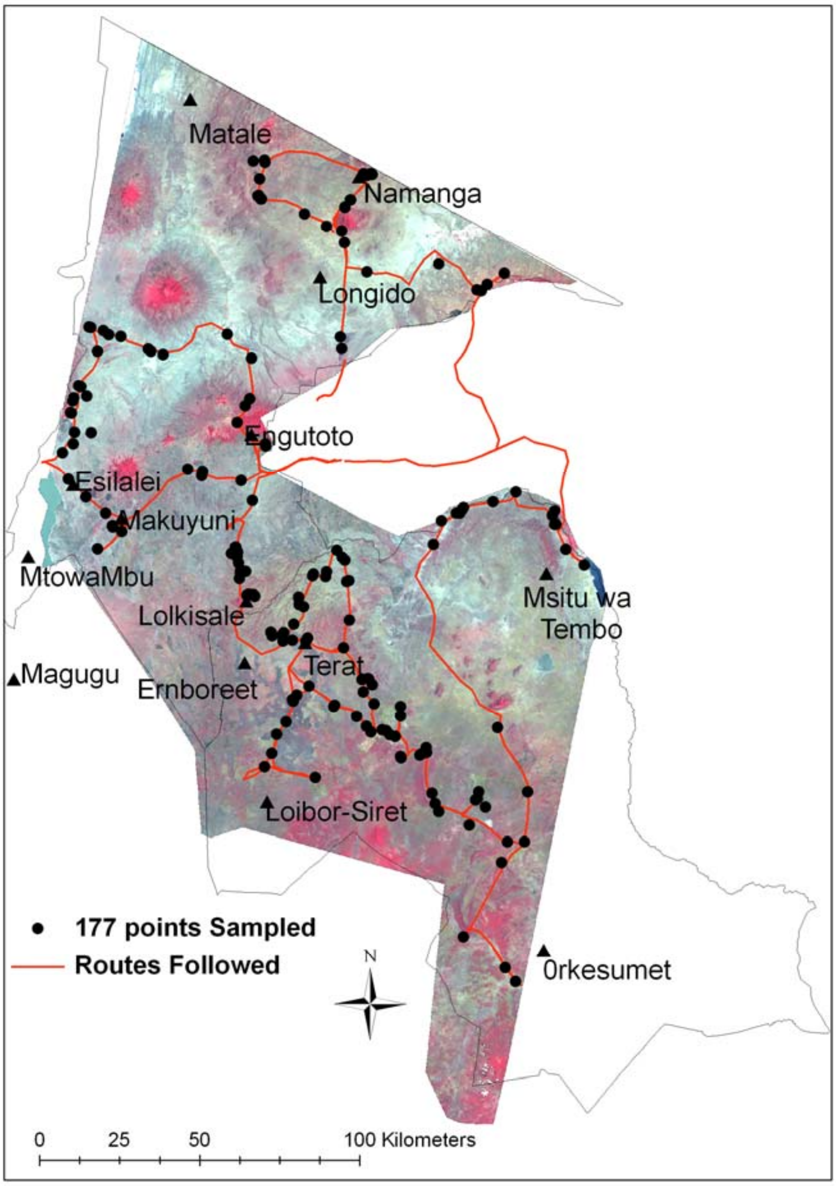

Figure 2B.6 Sampled points located near roads or paths in the study area

Table 2B.1 Data Form used in the field for ground verification and gathering historical information on land use and land cover changes

Village - Emboret- SIMA				1984				2000				2006-2007						
Ids	CHANGE	X_COORD	Y_COORD	Agric	Scale	Irrigate	crops	Agric	Scale	Irrigate	crops	Agric	Scale	Irrigate	Crops	When agr started	Photo	General comments
142	D	218922	9563147															
20	A	214080	9563000															
37	A	206553	9562745															
141	D	215065	9560455															
140	D	213809	9556149															
38	A	212804	9544246															
																		Change-code
	Codes:					**Crop types:**												**A- Agriculture 1984 None 2000**
	Agriculture	***Scale***	***Irrigated***			Maize - MZ	Others											B - No Agriculture 1984, Agric - 2000
	Agric - 1	Small - 1	Irrigated - 1			Beans - BN												C - Agric in 1984 and 2000
	Non Agric - 0	Large - 2	Non Irrig - 2			Ricce - RC												D - No Agricult 1984 and in 2000
						Wheat - Wh												

The coordinates and ids were uploaded into a Global Positioning System (GPS) was used to locate the Sample points. The following attribute value were picked upon locating the point: 1) Was there Agriculture in 1984, 2000 and 2007, 2) If small or Large scale, 3) Irrigated or not 4) When did Agriculture start, 5) Photographs were taken 6) Crop types 7) The change type picked during initial interpretation in the office was tested (A, B, C and D), 8) General comments. We did this with local Maasai elders who have the history of the area and also to help us interview locals a bout changes in Agriculture.

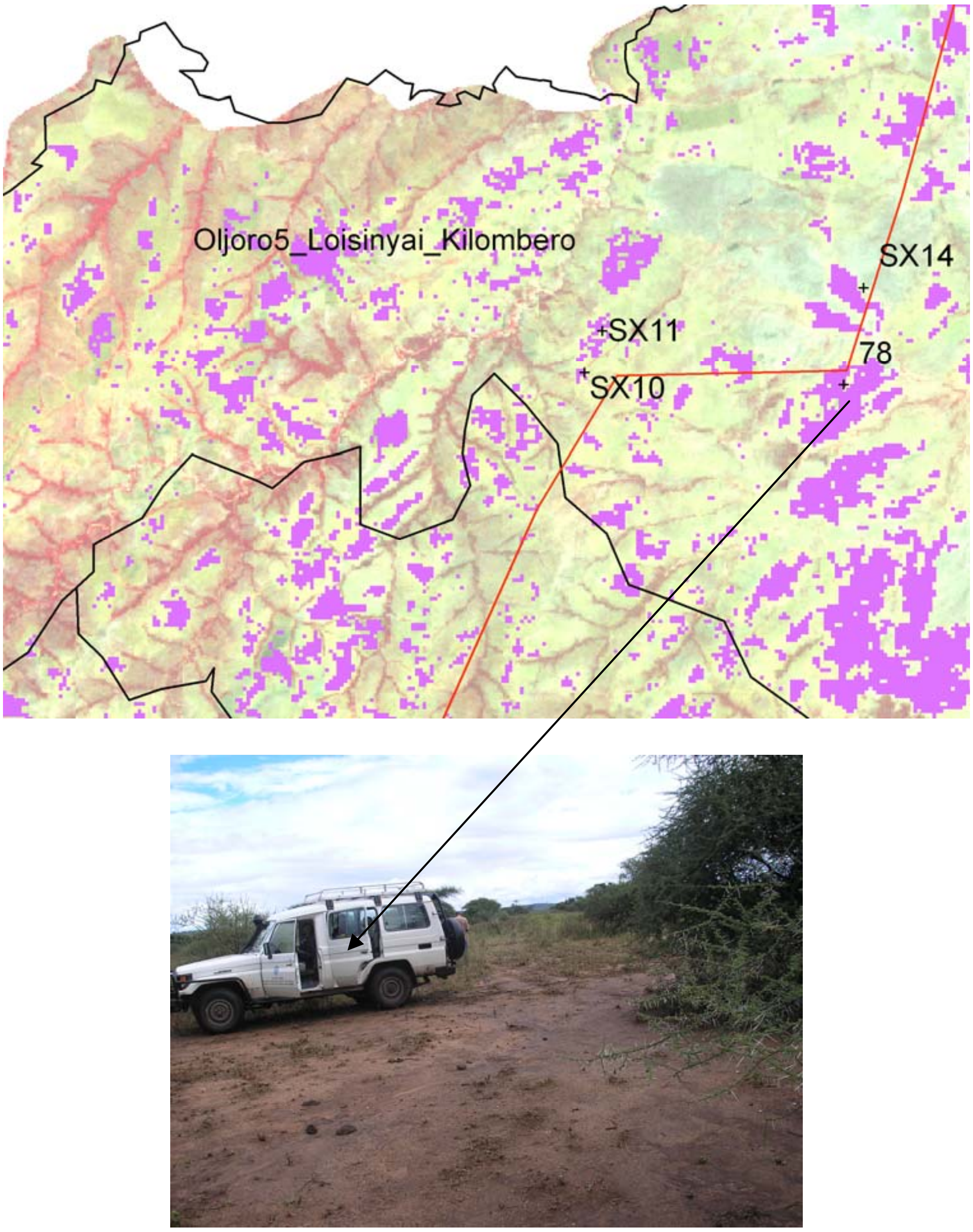

Figure 2B.7 Field verification of land-cover and land use changes in the Study area. The maps show Point 78 corresponds to the photograph- No Agriculture 1984, Started Late 1990's Non Irrigated, Small Scale, and Abandoned 2005.

After the field work new maps were generated. In order to evaluate and compare the classification quality a random-sampling accuracy assessment was conducted, comparing the classification results to a randomly chosen set of reference points. The following statistics were computed for each classification result, error matrix, overall accuracy, producer's accuracy, user's accuracy and Kappa statistics (Congalton & Green 1999). The overall accuracy of agriculture fields was about 99% with Kappa statistics of 0.98.

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