Direct Payments to Promote Biodiversity Conservation and the Implications for Poverty Reduction among Pastoral Communities in East African Arid and Semi-Arid Lands

Philip M. Osano
Department of Geography
McGill University
Montreal, Quebec, CANADA

A thesis submitted to McGill University in partial fulfillment of the requirements of the degree of
Doctor of Philosophy
July 2013

© Philip M. Osano, 2013

Abstract

Payments for ecosystem services (PES) are widely being applied in the conservation of biodiversity and management of ecosystem services. In developing countries, particularly in Africa, PES mechanisms are expected to generate co-benefits of poverty reduction and improved livelihoods in rural areas. However, there are very few such schemes in the Arid and Semi-Arid Lands (ASAL) that involve pastoral and semi-nomadic communities. Furthermore, there is little empirical evidence of the impact of PES on poverty. This thesis assesses the changes in biodiversity (using wildlife populations as a proxy), ecosystem services (provisioning, habitat and cultural and amenity services), human population and climatic variability in Kenyan ASAL at the national, sub-national and local levels. It further examines the potential and actual implications of PES on poverty and ecosystem based adaptation (EBA) to climate change among two pastoral Maasai communities living adjacent to wildlife protected areas in southern Kenya. It generates evidence of the positive effects of PES on the livelihoods of participating families, particularly during a period of extreme drought when PES serves as a critical safety-net against high livestock mortality and loss of cash income derived from livestock. In addition, PES is found to be an invaluable source of income diversification and is the most equitable of all income sources among participating households. Despite some of its positive attributes, there is need to assess and mitigate the potential negative implications of PES impact on the non-participating households and the landless families. Concerns also arise with regard to the equity implications, leakages and the lack of financial sustainability in the PES programs analysed.

Résumé

Les payements pour les services d'écosystème (PSE) sont largement utilisés pour la conservation de la biodiversité et le management des services environnementaux. Particulièrement en Afrique, les PSE sont supposés générer des revenus complémentaires pour la réduction de la pauvreté et l'amélioration des moyens de subsistance dans les zones rurales. Cependant, il y a très peu de PSE dans les zones arides et semi-arides (ZASA) qui impliquent les communautés pastorales et semi-nomades. Il y a peu d'évidences empiriques de l'impact des PSE sur la pauvreté. Cette thèse évalue les changements en biodiversité, les services environnementaux, la population humaine et la variation climatique dans les ZASA au Kenya. L'étude examine aussi les implications potentielles et réelles des PSE sur la pauvreté et les adaptations au changement climatique fondées sur les écosystèmes de deux communautés pastorales Maasai vivant à proximité des zones protégées de la faune sauvage dans le sud du Kenya. L'étude génère l'évidence des effets positifs des PSE sur les moyens de subsistance des familles participantes, particulièrement pendant une période d'extrême sécheresse quand les PSE servent comme un moyen de sécurité contre la mortalité élevée du bétail et la perte de revenus de l'élevage. Les PSE sont comme une source précieuse de diversification des revenus avec un effet de réduction de l'inégalité des revenus. Malgré ses aspects positifs, il est nécessaire d'évaluer et de contrôler les conséquences négatives potentielles de l'impact des PSE les non-participants et les familles sans terre. Les inquiétudes se posent sur les implications d'équité, et le manque de financement durable des programmes.

TABLE OF CONTENTS

List of Figures

List of Tables

Acknowledgements

I am extremely grateful to the numerous people who supported me throughout my PhD studies. My sincere appreciation goes to my Co-supervisors Prof. Peter G. Brown and Prof. John Galaty for the intellectual guidance that you provided to me all along, for your patience and constructive criticism of my work, and for always pointing me in the right direction when I seemed to waver. I thank Prof. Brown for accepting to serve as my Mentor when I first arrived at McGill in 2006 as part of the Sauvé Scholars Program, and for encouraging me to stay on for my PhD. I am indebted to Prof. Galaty for sharing with me the deep knowledge, and insights about the issues of pastoralism, wildlife conservation and land tenure change in East African rangelands.

I would also like to express my appreciation to the members of my PhD Advisory Committee, Prof. Thom Meredith of the Department of Geography and Dr. Markus Lehmann of the United Nations Secretariat of the Convention on Biological Diversity (UNCBD) for their intellectual guidance, stimulating ideas and unwavering support. Thank you to Prof. Tim Moore, Prof. George Wenzel, Maria Marcone, Ruilan Shi, Paula Kestelman and Anna Cerrone for the departmental support. I also acknowledge Shannon Scott of the McGill School of Environment (MSE) and Iain Blair of the McGill Institute for the Study of International Development (ISID) for facilitating access to my field research funding. My sincere gratitude goes to Professors Jon Unruh, Sarah Turner, Geraldine Akman and Eliane Ubalijoro for the encouragement at various times during my studies.

Generous funding for my doctoral studies and research came from several sources. At McGill, I was graciously funded by a Research Assistantship from Prof. Peter Brown provided through the McGill School of Environment (MSE), the J.W. McConnell Memorial Fellowship, and the Neil Croll Memorial Award. I am also grateful for the financial support towards my field research which I received from the Geography Department's Warren Development Geography Fund, the McGill Center for Development Areas Studies (CDAS), the International Development Research Center

(IDRC), the Africa Initiative Program of the Center for International Governance Innovation (CIGI), and the International Livestock Research Institute (ILRI). I also acknowledge additional support for thesis writing provided to me by Prof. Peter Brown and Prof. John Galaty and the African Technology Policy Studies Network (ATPS).

My life in Montreal would not have been the same without the support I received from Miriam Tees, Nancy and Jim Wright (RIP), and Prof. Mark and Christine Curtis all of whom welcomed me into their homes. Desiree McGraw, Diana Nicholson and the Sauvé Scholars were a great source of support and social network.

Permission to conduct research in Kenya was granted by the National Council for Science & Technology (NCST) in the Ministry of Higher Education, Science & Technology (MHEST). I also thank the Narok County Council, in particular, James Sindiyo, the Warden for the permission and access to the Maasai Mara National Reserve.

Several organizations and people contributed to and supported my research in Kenya. I am indebted to the International Livestock Research Institute (ILRI) for hosting me as a Graduate Fellow, and for availing data and facilities for my research and specifically to Dr Shirley Tarawali and Dr. Ian Wright for sanctioning my affiliation to ILRI. I am especially grateful to Dr. Mohammed Y. Said, my supervisor at ILRI, for sharing ideas, assisting me with data analysis, and for guiding me in my career. Thanks to Dr. Jan de Leeuw for stimulating ideas, guidance and support, and for the opportunities to be involved in ILRI's projects on payment for ecosystem services and climate change adaptation. My sincere gratitude to Shem Kifugo and Absalom Omolo for the assistance with GIS and the preparation of study maps, to Titus Karanja for support in developing the household survey database, and to Nicholas Ndiwa for broadening my understanding of statistics. Dr. Polly Ericksen, Dr. Silvia Silvestri, Leah Nganga, and Jane W. Gitau were extremely supportive, as was Duncan Turere. I would also like to acknowledge Dr. Kevin Urama for providing me with the opportunity to advance my research through the African Technology Policy Studies Network (ATPS) as a Researcher.

My research also involved collaborative work with others who deserve much credit. Thanks to Dr. Hippolyte Affognon for patiently guiding me through the economics of impact evaluation. Dr. Joseph Ogutu was at hand, always at very short notice, to help me with complex statistical problems. I learned a lot from working with Prof. Regina Birner and Sarah Schomers. My debt of gratitude also go to my student colleagues, both at McGill and at ILRI for the companionship, stimulating discussions, the endless debates and sharing of experiences. Geoffrey Garver, Alec Blair, Stephen Moiko, Xochitl (Itzel) Hernandez, Hellen Okochil, Archi Rastogi Claire Bedelian, and Caroline Bosire deserve mention. Many friends also supported me in one way or another. Thanks to Prof. Peter Blaze Corcoran, Prof. Akpezi Ogbuigwe, Mayor Morris Flewweling, and Robert Ouma.

My field research would not have been possible without the support of many people involved in the implementation of the PES programs that were the subject of my dissertation. In this regard, I am highly indebted to the present and past staff of The Wildlife Foundation (TWF); John Solonka, David Matiko, Irina Wandera, Ogeli Makui, Sylvia Sumare and Ed Loosli for providing me with the data on the Wildlife Lease Program. Dr. David Nkedianye provided useful background information concerning conservation in the Athi Kaputie Plains. In the Maasai Mara, I acknowledge the support provided by Rob O'Meara, James Kaigill, and John Saiyalel for availing data and sharing critical information about the Olare Orok Conservancy. Dickson Kaelo was instrumental in facilitating my fieldwork and providing critical background information on the development of Community Wildlife conservancies in the Maasai Mara Ecosystem. I am also indebted to Stanley Kimaren for taking me on my first trip to the Maasai Mara and for sharing insights on the social and governance challenges concerning wildlife tourism benefit sharing within the Mara Group Ranches.

I greatly appreciate the contributions of my able research assistants, Moses Kuyo Saitoti and Ntimama Matiany as well as my entire field research team at the two study sites who worked hard and contributed so much to this research: Kisham Makui, Wilson Ntaiya, Nickson Parmisa, Joseph Tuleto, Simon Peria, David Sorimpan, James Lenjir and Josephat Kirrokor. My greatest debt of all goes to the Maasai families in Athi Kaputie and Maasai Mara for their hospitality and sharing critical and valuable information for my research.

I would also like to thank my family for the role they have played in supporting my studies. Words cannot express how much I appreciate your support. You were a great source of strength during the time of my PhD research. Thank you to Nerreah, Clement, Olive, Justus, Ziadah and Dickens for your encouragement. To Gregory, Steve, Brian, and Julie, thanks for tirelessly running all my errands, and to Joy for all your support. My greatest gratitude of all belongs to my parents Gideon and Zipporah for their love, support and encouragement. Losing my mother only five days after returning to Canada in June 2010 was a great pain to bear, but I am sure she would have been as proud of my accomplishment, as she always was. Last and most important, I owe much to my wife Rahma for enduring my long absence. Indeed, I would not have made it this far without your unwavering support and endless prayers.

Contribution of the Authors

This thesis contains four manuscripts. Philip M. Osano is the first author in all the four manuscripts.

For Manuscript 1 (Chapter 3) — "*Evaluating the Potential of Payment for Ecosystem Services for Poverty Alleviation In Relation to Changes in Biodiversity and Ecosystem Services in Arid and Semi-Arid Lands in Kenya*", the first author was responsible for collection of data, analysis of data and writing of the manuscript. The second author, Caroline Bosire contributed to the compilation of the demographic data and of the database on Wildlife Conservancies in Kenyan Arid and Semi-Arid Lands (ASALs), and also provided editorial feedback. The third author, Mohammed Y. Said and the fourth author, Shem Kifugo both contributed to data collection and GIS based analysis and prepared four figures for the final manuscript. The last author Jan de Leeuw contributed to statistical analysis, writing of sections of the manuscript and provided editorial feedback.

For Manuscript 2 (Chapter 4) — "*Exploring the Role of Payments for Environmental Services in Ecosystem Based Adaptation to Climate Change and Poverty Alleviation: Insights from Kenyan Rangelands*", the first author was responsible for collection of data, analysis of data and writing of the manuscript. The second author, Joseph O. Ogutu developed the drought assessment model and prepared two figures for the final manuscript.

For Manuscript 3 (Chapter 5) — "*Poverty, Inequality and Participation of Pastoralists in a Payment for Ecosystem Service Scheme Adjacent to a Semi-Arid Protected Area in Southern Kenya*", the first author was responsible for design and collection of the household survey data, analysis of data and writing of the manuscript. The second author Mohammed Y. Said contributed to the design of the household survey, data collection, spatial analysis in GIS (together with the third author), and prepared two figures for final manuscript. The third author, Shem Kifugo generated the spatial data and contributed to GIS based analysis and preparation of two figures. The fourth author Jan de Leeuw contributed to statistical analysis, and provided inputs to the manuscript write up. The fifth author Nicholas Ndiwa provided ideas and helped in testing different tools and statistical methods and contributed to the computation of a household asset index using Principal Component Analysis (PCA). The sixth author Hippolyte Affognon contributed to the poverty impact analysis, and developed the two econometric models. The last author Joseph O. Ogutu reviewed and modified one of the econometric models prepared by the sixth author, developed a model to calculate the differences in gini-index between the two sub-samples of the survey population, and reviewed and provided editorial feedback on the final manuscript.

For Manuscript 4 (Chapter 6) **—**"*Why Keep Lions Instead of Livestock? An Assessment of the Effect of Wildlife Tourism- Based Payment for Ecosystem Services on Herders in the Maasai Mara Ecosystem*, *Kenya*", the first author was responsible for the design and collection of the household survey data, analysis of data and the writing of the manuscript. The second author Mohammed Y. Said provided guidance on the design of the household survey, and conducted spatial analysis in GIS. The third author Jan de Leeuw contributed to the design of the household survey, ideas for analysis, and provided feedback on the manuscript. The fourth author Nicholas Ndiwa contributed to the regression analyses. The fifth author Dickson Kaelo provided data on the Olare Orok Conservancy, the background information on the study site and helped to facilitate field data collection and focus group discussions. The sixth author Sarah Schomers led a SWOT (Strengths, Weaknesses Opportunities and Threats) analysis of Conservancies. The seventh author Regina Birner led a participatory mapping exercise using Net-map (Influence-Network-Mapping) with representatives of tourism partners in two Conservancies in the Maasai Mara Ecosystem to assess the institutional and governance arrangements. The last author, Joseph O. Ogutu developed a model to calculate the differences in gini-index between the two sub-samples of the survey population, and reviewed and provided editorial feedbacks.

CHAPTER 1: INTRODUCTION

In this thesis, I assess the implications of direct payment for biodiversity, a form of payment for ecosystem services (PES), specifically focused on wildlife conservation, on the poverty conditions of pastoral and agro-pastoral communities in the Arid and Semi-Arid Lands (ASALs) in East Africa. Consideration is placed on wildlife biomass and populations as a proxy for biodiversity within the context of land use management at the livestock-wildlife interface in the ASALs (Du Toit et al., 2010). This study therefore contributes to the growing literature regarding the linkages between PES and poverty in the developing countries (Bulte et al., 2008b, Ravnborg et al., 2007).

Much of the existing literature on this subject is concerned with PES programs that target hydrological services for watershed conservation and carbon sequestration for climate change mitigation mostly in forest based ecosystems and in mixed agricultural landscapes (FAO, 2007, Lipper et al., 2009, Pagiola et al., 2005). To date, there is very little documentation of PES experiences in the ASAL (dryland) ecosystems, including in the rangelands. This thesis therefore contributes to filling the critical knowledge gap that currently exists with regard to PES implementation in rangelands and how it affects nomadic and semi-nomadic pastoral communities in sub-Saharan Africa (Dougill et al., 2012).

Biodiversity conservation and poverty reduction are both major challenges at the global, national and local levels. Policy makers, scholars and practitioners in both areas are working to create conceptual and practical links, with the ultimate goal to ensure improved human welfare conditions without degrading the ecological base which supports life on earth (Sachs et al., 2009). Although some scholars consider PES as purely an environmental policy mechanism, there is considerable debate about whether PES should, and to what extent it can contribute to poverty reduction at all levels, from the global, national and local levels to the household unit.

Two schools of thought have emerged in this unresolved debate. One school, although recognizing the poverty alleviation potential of PES, rejects the call to explicitly include poverty reduction as a principal objective in the design and implementation of PES programs (Pagiola, 2007). This school considers the integration of poverty alleviation objectives into PES schemes as an "unnecessary burden" (Engel et al., 2008, Pagiola et al., 2005). Proponents of this school argue that poverty reduction can emerge as a "side-benefit" or co-benefit of PES programs (Pagiola, 2007).

An alternative pro-poor school argues that PES should not be so narrowly conceptualized as to exclude explicit poverty reduction and equity objectives in its design and implementation (Pascual et al., 2010). This school recognizes that the conservation of ecosystem services and poverty reduction are inseparable aspects of sustainable development. Consequently, even if a PES program is not designed with explicit poverty reduction objectives, its implementation will necessarily have implications on the poverty status of both PES participants and non-participants alike (Bulte et al., 2008b).

A major issue in the context of PES interventions within the biodiversity-poverty nexus to recognize that any intervention, whether for poverty reduction or to promote improved environmental management will generate impacts; "*...even if a change in program or policy has a single objective, be it social or environmental, we must recognize that the impacts may be felt by both the human population and the ecosystems in which they live*" (Barrett et al., 2011). In this thesis, I approach the PES-Poverty debate from the view that irrespective of the school of thought that one adopts, it is still critical to scrutinize PES programs regarding their implications for and the effects on poverty among the different stakeholder groups.

While the theory on the different dimensions of linkages between PES and poverty has advanced considerably (Bulte et al., 2008b, Zilberman et al., 2008, Pagiola et al., 2005, Muradian et al., 2010), in practice, there are only few actual initiatives and even fewer studies on the impact of PES on poverty in the developing world. This lack of empirical studies and dearth of evidence is especially pertinent in the dryland ecosystems, especially the ASALs, which have not particularly been attractive for PES implementation. Thus, we know very little about the participation of semi-nomadic pastoral communities in PES programs, let alone about the poverty impact of PES among these groups.

In this thesis, I begin to address this knowledge gap drawing on case studies involving the Maasai community, considered among the poorest and most economically vulnerable in East Africa (Homewood et al., 2009c). High levels of income poverty among the Maasai in East Africa persists despite their occupying some of the richest areas in terms of the diversity and populations of wildlife species that generate high tourism revenues (Homewood et al., 2012). Although there have been several studies concerning the poverty outcomes of wildlife based interventions such as tourism, there has so far not been any study that looks at PES programs implemented through direct payments to pastoral households.

Thesis objectives

This thesis has two main objectives. The first objective is to assess the impact of PES targeting biodiversity services on the poor pastoral communities living around wildlife protected areas. Three of the four research chapters are based on the case studies of PES schemes that involve pastoral communities living adjacent to the Nairobi National Park (NNP), and the Maasai Mara National Reserve (MMNR) in southern Kenya. The second objective is to examine the role of PES as mechanism to support ecosystem-based adaptation (EBA) to climate change and poverty reduction in the context of drought risk mitigation among pastoral communities in the ASAL areas.

Thesis outline

This chapter gives the introduction to the thesis. It consists of the thesis objectives, and the thesis outline, which includes information on the other six chapters, their sequence in the thesis and a brief overview of the chapter contents.

Chapter two provides a literature review of a number of concepts and issues that relate to the topic of the thesis. These include the definition and review of literature on biodiversity, ecosystem services, poverty, climate variability and change, and the linkages among them. Critical gaps in the scientific literature concerning these research topics are also identified. I conclude the chapter by presenting the criteria for the selection of the two case study sites and PES programs presented in chapters 4, 5 and 6.

Chapter three is the first manuscript of the thesis (Osano et al., submitted to *Environmental Development*). It explores the potential for wildlife Payments for Ecosystem Services (PES) to alleviate poverty in relation to changes in biodiversity (using absolute wildlife biomass as a proxy), ecosystem services, land use and demography for the 30 year period between 1970s and 2000s, and climate variability in the Kenyan ASALs. In the chapter, two provisioning ecosystem services; crop and livestock production, together with habitat services are analyzed for the entire Kenyan ASAL. Again, two ecosystem services; habitat services and cultural services of tourism, are analyzed for Maasailand, the ASAL parts of southern Kenya that is predominantly occupied by the Maasai communities.

Chapter four is the second manuscript of the thesis. It is an updated version of a manuscript published in an edited book volume titled; "*Africa Rising: A Continent's Future Through the Eyes of Emerging Scholars*"(Shaw and Mackinnon, 2013) . The ideas addressed in this chapter emerged following the 2008-2009 drought that coincided with my fieldwork period. Witnessing firsthand how the Maasai pastoral families struggled to cope with the effects of this devastating drought (Osano, 2011), informed my decision to the potential of ecosystem-based adaptation (EBA) to climate change. I specifically examined the role of payments for environmental services (PES) and its potential in drought-coping among pastoral communities.

In this chapter I first present a conceptual framework of the inter-linkages between PES and EBA, and then analyze the frequency, occurrence and severity of drought in the last 40 and 90 years in Athi-Kaputie Plains and the Maasai Mara Ecosystem respectively.

This is then followed by an assessment of the effects of PES on the *adaptive capacity* of the PES participants, and on the local institutions relevant to climate change adaptation. In relation to the *adaptive capacity* of the PES participants, I evaluate the PES effects on three determinants of adaptation; the economic assets and wealth; human capital, access to technology and infrastructure; and empowerment and local governance. With respect to institutions relevant to adaptation at the local level, I identify and discuss three effects of PES, namely the establishment of new land use and land management rules and regulations; the interactions affecting collective action institutions that can support or undermine local adaptation responses and coping strategies; and the PES effects on both the inter-sectoral and cross-sectoral linkages.

Chapter five is the third manuscript of the thesis (Osano et al., to be submitted to *Ecological Economics*). It is based on research conducted in the study site in Athi-Kaputie Plains (AKP) which is located to the south of Nairobi National Park (NNP) in south-eastern Kenya. I conduct an assessment of the Wildlife Lease Program (WLP), a PES program whereby pastoral land owners are paid US$ 10/ha/year to refrain from cultivation, land sales and sub-division, and to allow wildlife on their private land.
I assess the programs effects on poverty and inequality among participating and non-participating households. I also examine the programs institutional arrangement based on its design and implementation, the level of household poverty (based on cash income, land ownership and a composite household asset index) and wealth inequality (in terms of cash income, livestock holdings and land ownership) among the target households, the determinants of participation and of the intensity of participation in the WLP as well as the effects of the PES on household poverty, inequality, income and expenditure.

Chapter six is the last manuscript of the thesis (Osano et al., resubmitted to *Natural Resources Forum*). It is based on the research conducted in the Maasai Mara Ecosystem (MME) study site which is located in south-western Kenya. I examine the effects of a Payment for Ecosystem Services (PES) program on household poverty, wealth inequality and pastoral livelihoods under changing land tenure, using the case of the

Olare Orok Conservancy (OOC). In this PES scheme, participating Maasai landowners have agreed to voluntary resettlement and exclusion of livestock grazing by collectively enrolling their land in the conservancy which is set aside for wildlife tourism, in return for being paid US$ 41/ha/year (in 2011) by a coalition of private sector based commercial tourism operators. I examine the evolution of land tenure and the household settlement patterns in the MME between 1959 and 2012, the institutional arrangements in the design and implementation of the OOC PES program, the level of poverty (based on cash income and livestock holdings) and wealth inequality (in terms of cash income, livestock holdings and land ownership) among households in the study area, and the effects of the PES on household poverty, inequality, income and expenditure.

Chapter 7 is the conclusion. The chapter is divided into four parts. The first part is a summary of the results and findings of each of the four substantive thesis chapters. The second part situates the study findings in light of the debate on the ethical issues concerning PES and the Neo-Classical Economic (NCE) framework. The third part is a synthesis of the four main conclusions drawn from this study. It includes a discussion of the conclusion in relation to the literature on PES and poverty and how this plays out in the context of semi-nomadic pastoral communities. The last part provides an overview of some of the study limitations. It also includes a retrospective reflection on this study and provides suggestions on what and how these limitations and other gaps in the study would have been addressed. It concludes by identifying pertinent research areas for the future.

CHAPTER 2: LITERATURE REVIEW AND CASE STUDY SELECTION

This chapter consists of two sections. The first section covers the literature review, and the second covers the case study selection. At the end, a summary of questions to be addressed in this thesis is provided. In the first section of this chapter, I provide a literature review of a number of concepts and issues that relate to the topic of the thesis. These include the definition of and the review of literature on biodiversity, ecosystem services, climate change and variability, and poverty, and the linkages among them. Biodiversity underpins ecosystem services and is therefore the focus of conservation interventions, given its degradation and loss in current contexts of land use and climatic changes.

Ecosystems services are considered as the benefits that humans derive from ecosystems, which is highly dependent on the quality of biodiversity. the key question posed by the thesis is whether ‘payments’ to pastoral land users in ASALs made in exchange for the conservation of biodiversity (mainly wildlife) to support the provision of ecosystem services will provide adequate incentives to shape conservation behavior and to generate meaningful impact on poverty. Critical gaps in the scientific literature concerning the research topic of this thesis are also identified. The chapter concludes by presenting the criteria for the selection of the two case study sites and PES programs evaluated in subsequent chapters.

Biodiversity and ecosystems services

Defining biodiversity

Biodiversity is shorthand for “biological diversity”, and it has diverse definitions depending on the interpretations involved. These definitions deal with different organizational levels (genetic, species, ecosystems), different types of ecosystems and species (wild and domesticated), different spatial scales and one or both of the key elements ‘richness’ and ‘abundance’(Purvis and Hector, 2000). The most widely used definition is that by the United Nations Convention on Biological Diversity (UNCBD) which defines biodiversity as the;

> "Variability among living organisms from all sources including, inter-alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and ecosystems"(Secretariat of the Convention on Biological Diversity, 2010a).

Biodiversity has three dimensions within which variability occur: *genetic*, meaning the variation of genes within a species, sub-species or population; *population/species*, meaning the variation between living species and their component populations at different spatial scales (local, regional or global); and *community/ecosystem*, meaning the variation within ecological complexes of which species are a part. The management of biodiversity at landscape level mostly concerns populations and ecosystems linked to ecosystem services (Balvanera et al., 2001).

Classification of ecosystems

There are a number of different classifications for ecosystems, two of which are highlighted here; the Millennium Ecosystem Assessment (MEA) and The Economics of Ecosystems and Biodiversity (TEEB). The MEA used 10 ecosystem categories to report its findings. All the ecosystems within each category share a suite of biological, climate and social factors that tend to differ across categories. The MEA ecosystem category that is relevant for this study is "dryland ecosystems", which represent areas where plant production is limited by water availability and are defined by an aridity index value (the ratio of actual to potential evapo-transpiration) of less than 0.65 (Safriel and Adeel, 2005).

The dominant land uses in dryland ecosystem support grazing for the large mammal wildlife and domestic herbivores. There are four sub-types of dryland ecosystem that differ in the degree of water limitation and increasing level of aridity: (1) the dry sub-humid; (2) semiarid; (3) arid; and (4) hyper-arid. These dryland sub-types are sometimes described in terms of their land uses such as rangelands, croplands and urban areas (Safriel and Adeel, 2005). This thesis focuses on the Arid and Semi-Arid Lands (ASALs) sub-type of dryland ecosystem.

The TEEB presents two levels of ecosystem classification. Level 1 classification is based on 12 biomes, which are further sub-divided into Level 2 classifications. The latter consist of 29 ecosystems. The level 1 classification relevant for this thesis is the "Grass/rangeland biome", with the principal ecosystem of interest within it being the savannah grasslands (TEEB, 2010).

Defining ecosystem services and environmental services

In the literature, the terminologies '*ecosystem services'* and '*environmental services*' are sometimes used interchangeably. There is however some differences between these two terminologies (Muradian et al., 2010). I first highlight the definition of "ecosystem services", before pointing out how this definition differs from that of "environmental services" as conceptualized by different authors. The literature is now is abound with multiple definitions of "ecosystem services" and seven of these are described in Table 2.1. These alternative definitions of "ecosystem services" have evolved over time, and the concept is still evolving with varying attention to the ecological basis or economic use (Braat and de Groot, 2012).

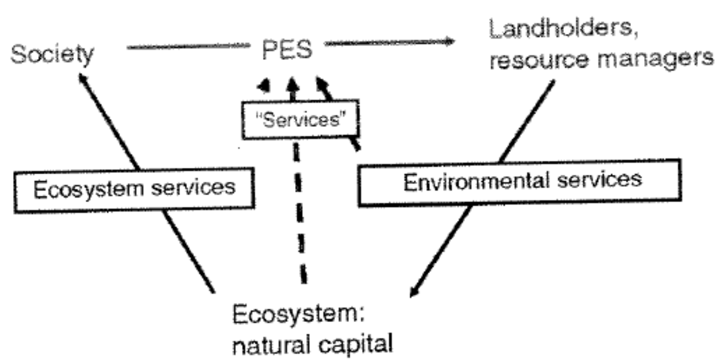
Table 2.1 The different definitions of "ecosystem services" in the literature
Source; Braat & de Groot (2012).

Author	**Definition of Ecosystem Services**
(Daily, 1997)	*Ecosystem Services* are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life
(Costanza et al., 1997)	*Ecosystem Services* are the benefits human populations derive, directly or indirectly, from ecosystem functions
(Millennium Ecosystem Assessment, 2005b)	*Ecosystem Services* are the benefits people obtain from ecosystems
(Boyd and Banzhaf, 2007)	*Ecosystem Services* are components of nature, directly enjoyed, consumed, or used to yield human well-being
(Fisher et al., 2009)	*Ecosystem Services* are the aspects of ecosystems utilized (actively or passively) to produce human well-being
(TEEB, 2010)	*Ecosystem Services* are the direct and in direct contributions of ecosystems to human well-being

Some authors make a distinction in the meaning between the terms 'ecosystem services' and 'environmental' services. Swallow et al (2009) consider the inclusion or exclusion of *provisioning* ecosystem services as the main difference between "ecosystem services" and "environmental services". According to these authors, the *provisioning* ecosystem services as classified by the Millennium Ecosystem Assessment such as food, fiber, and timber for example, are excludable and non-competitive goods for which markets develop most readily. Swallow et al., thus consider "environmental services" as excluding the *provisioning* services, but mainly incorporating the regulating, supporting, and cultural services of ecosystems for which markets do not easily develop (Swallow et al., 2009).

A different perspective is provided by Greiner and colleagues, who argue that the conceptual distinction between ecosystem and environmental services lies in the fact that ecosystem services are derived from natural capital while environmental services are provided by actors. As shown in Figure 2.1, the notion of 'ecosystem services' is understood in this context to be outcome based, and focus on the well-being benefits provided to society from natural capital. In contrast, the notion of 'environmental services' is input based and focuses on the efforts undertaken by actors to generate environmental improvements and improved natural capital (Greiner et al., 2009).

Figure 2.1 Conceptualization of ecosystem and environmental services.
Source; Greiner et al. (2009, p. 54).



Except for chapter four of this thesis, I mainly use the term 'ecosystem services' based on the conceptualisation that considers 'ecosystem services' as a sub-category of '*environmental services that deal exclusively with the human benefits derived from natural ecosystems*'. In this context, the term 'environmental services' which is used in chapter 4 is much broader and encompasses the benefits associated with different types of actively managed ecosystems, including agricultural practices and rural landscapes (Muradian et al., 2010).

Classifying ecosystem services

The Millennium Ecosystem Assessment (MEA) identified four different categories of ecosystem services. These are *provisioning services* such as food, water, timber, and fiber; *regulating services* that affect climate, floods, disease, wastes, and water quality; *cultural services* that provide recreational, aesthetic, and spiritual benefits; and *supporting services* such as soil formation, photosynthesis, and nutrient cycling (Millennium Ecosystem Assessment, 2005b).

The Economics of Ecosystem and Biodiversity (TEEB) presents a typology of 22 ecosystem services which are classified in four different categories; provisioning services, regulating services, habitat services, and cultural and amenity services (TEEB, 2010). The TEEB classifies "habitat services" as a distinct category to highlight two features: a) the interconnectedness of ecosystems in the sense that different ecosystems provide unique and crucial habitats for particular life-cycle stages of migratory species; and b) that certain ecosystems have been identified that are of unique importance in that they have been found to exhibit particularly high levels of genetic diversity of major importance to maintain life (genetic diversity) on Earth, and natural adaptation processes. Both of these features underpin all, or most, provisioning, regulating and cultural services (TEEB, 2010).

There are two critical differences between the MEA and the TEEB definition and classification of ecosystem services. The first difference is that although the TEEB definition is based on that of MEA, it makes a finer distinction between 'services' and 'benefits' which are seen as distinct and are therefore not identical. This differences is elucidated in detail by distinguishing the intermediate and final ecosystem services and the benefits these generate to humanity to support human well-being (Fisher et al., 2009). The second major difference is that unlike in the MEA, the TEEB classification excludes “supporting services”. In the TEEB, supporting services are considered as a sub-set of ecological processes that are encapsulated in the ‘habitat services” and which takes into account the importance of ecosystems to provide habitat for migratory species and as ‘gene-pool protectors’ (e.g. natural habitats allowing natural processes to maintain the vitality of the gene pool) (TEEB, 2010). I have chosen to use the TEEB definition and classification in this thesis because I consider “habitat services” as providing an ideal description of pastoralists land use allocation for wildlife grazing through PES.

Responses to the loss of biodiversity and the degradation of ecosystem services

Biodiversity plays an important role in ecosystem functions that provide ecosystem services but recent assessments show significant declines globally and across different ecosystems (wetlands, forests, rangelands etc). The loss of biodiversity also contributes to the degradation of ecosystem services, and is attributed to both direct and indirect drivers of change (Millennium Ecosystem Assessment, 2005a). The five direct drivers of biodiversity loss and ecosystem change include: (1) land use change; (2) climate change; (3) spread of invasive species; (4) overexploitation; and (5) pollution. The indirect drivers are varied, and include demographic changes, economic factors, socio-political factors, cultural factors, and technological factors (Secretariat of the Convention on Biological Diversity, 2010a).

A number of tools and conservation mechanisms have been developed in response to the loss of biodiversity and degradation of ecosystem services. These tools, mechanisms and approaches are applied widely to promote sustainable use and other measures for protection of biodiversity. These include protected areas, ecosystem restoration, legislation at national and international levels to regulate the harvesting of wildlife species or to establish penalties for land use practices considered to be harmful to biodiversity and an array of integrated conservation and development projects (ICDPs).

According to the MEA, while many of these mechanisms have been successful in limiting biodiversity loss and homogenization to rates lower than they would otherwise have been in the absence of such mechanisms, they remain inadequate. Thus, a portfolio of actions that build on current initiatives are therefore needed to address the direct and indirect drivers of biodiversity loss and ecosystem service degradation (Millennium Ecosystem Assessment, 2005a). This thesis considers one such mechanism, payments for ecosystem services (PES) which also includes Markets for Ecosystem Services (MES) if these are based on private sector arrangements (Jenkins et al., 2004, OECD, 2004, OECD, 2010). The thesis specifically focuses on the direct payment for biodiversity initiatives (Ferraro and Kiss, 2002, Milne and Niesten, 2009) as applied in the ASAL rangelands.

Poverty and human well-being

The concept and definition of poverty

The concept of poverty has evolved from approaches that are based on the ideas of *subsistence*, to *basic needs* and more recently to *relative deprivation* (Townsend, 2006). The *subsistence approach* to poverty is based on minimum necessities required to meet the nutritional needs of an individual to fulfill physical survival and efficiency (DeRose et al., 1998). The *basic needs approach* to poverty is an extension of the subsistence approach, with addition of two new elements: a minimum consumption needs of a family (adequate food, shelter, clothing and certain household furniture); and essential services provided by and for the community (Sachs, 2005).

The basic needs approach therefore extends the concept of poverty from a focus on an individual as reflected in the subsistence needs approach, to a focus on the household (family) and community (Townsend, 2006). The *relative deprivation approach* has expanded the conceptualization of poverty from the narrow focus on financial income to a broader multi-dimensional perspective which considers poverty as material deprivation, lack of access to basic needs (food, education, health etc), the absence of political autonomy, lack of freedom of choice, and socio-economic inequality (Alkire and Foster, 2011, Sen, 1999).

This multi-dimensional conception of poverty was advanced by Amartya Sen who proposed the following five instrumental freedoms without which one would be considered poor: economic facility; social opportunity; security; transparency; and political freedom (Sen, 1999). The relative deprivation concept can be closely related to that of *human well-being,* which also has multiple constituents, including basic material for a good life, freedom of choice and action, health, good social relations, and security. Well-being is at the opposite end of a continuum from poverty, which has been defined as a ‘‘*pronounced deprivation in well-being*’’(Millennium Ecosystem Assessment, 2005b).

Some international development agencies dealing with poverty in the developing countries recognize and in some cases have also adopted the multi-dimensional perspective of poverty, as a complement to the traditional measure based on income. The United Nations Development Programme (UNDP) introduced the Human Poverty Index (HPI), which is measure of poverty that incorporates two of the five instrumental freedoms proposed by Amartya Sen; economic facility and social opportunity (UNDP, 1997). The Development Assistance Committee of the Organization of Economic Cooperation and Development (DAC-OECD) has conceptualized poverty as consisting of five key dimensions. These are economic (e.g. income, decent work), human (e.g. health, education), socio-cultural (e.g. status, dignity), and protective (e.g. insecurity, risk, vulnerability) dimensions (OECD-DAC, 2001). Although much of its policy work on poverty is still largely based on the conventional income definition, the World Bank has also previously in its 2000/01 World Development Report conceptualized poverty more broadly to include three dimensions namely opportunity; empowerment and security (World Bank, 2001).

Measurement of poverty

There is currently no single commonly agreed way to measure poverty largely due to its multi-dimensional nature. A monetary metric is currently the standard measure of defining income poverty in the developing world (Chen and Ravallion, 2007), and arbitrarily chosen international poverty lines of a per capita income of US$ 1.25 and 2 per day (in 1993 purchasing power parity) is accepted as the global benchmark for poverty (Deaton, 2003). To capture fully the multi-dimensional aspect of poverty, qualitative and quantitative non-income indicators have also been developed (Coudouel et al., 2002). These include for example indicators such as access to education, child mortality, malnutrition, and other asset based indicators (Alkire and Foster, 2011, Coudouel et al., 2002).

In this thesis, I move beyond the conventional measure of poverty based on monetary income to also consider and assess poverty based on household assets such as land, livestock and a composite asset index (CAI). The use of non-monetary measures of poverty is particularly useful in this study because the research subjects are drawn from a semi-nomadic community that consider livestock assets as a more meaningful and relevant indicator of poverty and wealth status (Tache and Sjaastad, 2010). Thus, the approach taken in this study also helps to address the shortcomings of using income based definition to classify pastoral households as poor yet, as clearly espoused in the book: "The Poor Are Not Us: Poverty and Pastoralism in East Africa" (Anderson and Broch-Due, 1999), the majority of pastoral households do not view or consider themselves as poor simply because they have low incomes.

Addressing the challenge of poverty: alleviation, reduction or prevention?
When dealing with the challenge of poverty, there is a distinct difference among the three possible outcomes of poverty interventions. It is thus critical to be clear and to understand whether the outcome of an intervention results in poverty alleviation, poverty reduction or poverty prevention because these are different outcomes altogether. "*Poverty alleviation*" refers to the situation where an intervention addresses some symptoms of poverty but does not actually lead to the transformation of poor people to non-poor. "*Poverty reduction*" implies that all poor people are lifted beyond a defined poverty line thereby transforming them from poor to non-poor status. Lastly, "*Poverty prevention*" refers to a situation whereby an intervention leads to preventing people from falling below the poverty line, or deeper into poverty if they were already

living below the poverty line (King and Palmer, 2007). These differences are fundamental in understanding the effects of an intervention or mechanism on poverty.

Poverty in relation to biodiversity and environmental change

Since the United Nations Conference on Human Environment (UNCHE) held in Stockholm in 1972, a near-universal consensus has developed that poverty and environment are inextricably linked. Efforts to mainstream environmental issues in development planning at national and global levels expanded rapidly following the 1987 publication of the widely acclaimed report, "Our Common Future" (WCED, 1987). This landmark report attributed the connection between poverty and environment thus;

> "Poverty is a major cause and effect of global environmental problems. It is therefore futile to attempt to deal with environmental problems without a broader perspective that encompasses the factors underlying world poverty and international inequality"
> (WCED, 1987, p.3)

A later review of evidence of the linkages between poverty and environment rejected this simple causal link finding the poverty-environment nexus as governed by a "complex web of factors" (Duraiappah, 1998), with variations based on the local socio-economic and also larger macro-economic policy context (Durning, 1989, Reardon and Vosti, 1995). Additional studies showed the potential for successfully combining poverty alleviation with environmental management interventions to realize positive outcomes for both (Ekbom and Bojo, 1999).

Since the early 1990s, policy makers and practitioners have intensified the integration of environment and development in pursuit of "sustainable development". Core to these efforts now include the integration of biodiversity conservation and poverty reduction have since became a central part of this agenda (Kishore, 2007, Pearce, 2005, World Resources Institute, 2005).

Although the links between environment and development have been of focus since early 1970s, that between biodiversity (as a component of environment) and poverty (as a component of development) have only recently gained serious policy and scholarly attention (Roe, 2010, Secretariat of the Convention on Biological Diversity, 2010b, Tekelenburg et al., 2009). This helps partially to explain the current dearth of empirical evidence regarding poverty-biodiversity linkages that has been bemoaned by scholars (Barrett et al., 2011, Tekelenburg et al., 2009).

In the biodiversity conservation arena, the United Nations Convention on Biological Diversity (UNCBD) provides the policy framework to link biodiversity to poverty reduction at the global and national level. The UNCBD acknowledges the relationship of biodiversity to poverty reduction, and states in its Preamble that;

> "Economic and social development and poverty eradication are the first and overriding priorities of developing countries" (www.cbd.int).

Since its inception in 1992, the UNCBD has promoted the integration of poverty reduction in international and national biodiversity conservation strategies. Most recently, the CBD 2010 target agreed upon in April 2002, explicitly mentioned poverty alleviation, and committed the Parties to the convention to;

> '...achieve by 2010 a significant reduction of the current rate of biodiversity loss at the global, regional and national level as a contribution to poverty alleviation and the benefit of all life on Earth' (Secretariat of the Convention on Biological Diversity, 2010a).

In the development arena, the Millennium Development Goals (MDGs) provides a framework to link poverty reduction to biodiversity conservation. MDG 7 includes a target to "*reverse the loss of environmental resources*", and one of the indicator developed in 2001 is the area of land under protection for biodiversity. In 2006, the MDG 7 was updated to include the CBD "2010 target" with additional biodiversity indicators (Sachs et al., 2009).

There are multiple frameworks on the linkages between poverty reduction on one hand and biodiversity and ecosystem services on the other (Fisher et al., In press). However, much attention has been paid to the frameworks developed in the MEA and the TEEB assessments. The MEA framework is very explicit regarding the inter-linkages among biodiversity, ecosystem services and human wellbeing and poverty reduction (Figure

2.2). In this framework, biodiversity is considered as underpinning the delivery of a range of ecosystem goods and services on which human well-being depends and poverty in this context is viewed as "*the pronounced deprivation of well-being*" (Millennium Ecosystem Assessment 2005a, p 29).

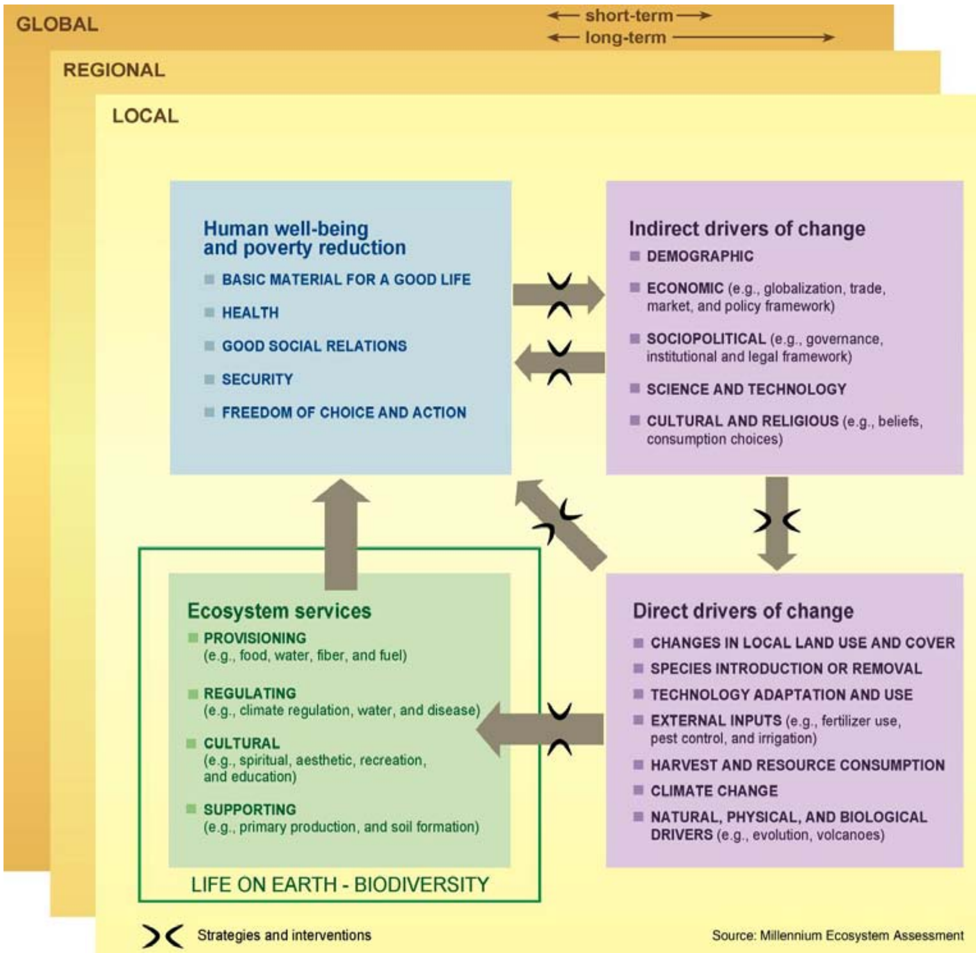
The MEA framework recognizes seven direct drivers of change that operate across local, regional and global scales in both the short and long term basis. These direct drivers include;

- Changes in local land use and cover
- Species introduction or removal
- Technology adaptation and use
- External inputs such as fertilizer use and irrigation
- Harvest and resource consumption
- Climate change
- Natural, physical and biological drivers such as evolution and volcanoes.

The human well-being and poverty are conceptualized more broadly in terms of five aspects that conform to the multi-dimensional nature of poverty. These aspects are (Millennium Ecosystem Assessment, 2005b):

- Basic material for a good life
- Health
- Good social relations
- Security
- Freedom of choice and action

Figure 2.2 A conceptual framework of interactions among biodiversity, ecosystem services, human well-being (including poverty) and the direct and indirect drivers of change.
Source; (Millennium Ecosystem Assessment, 2005b).

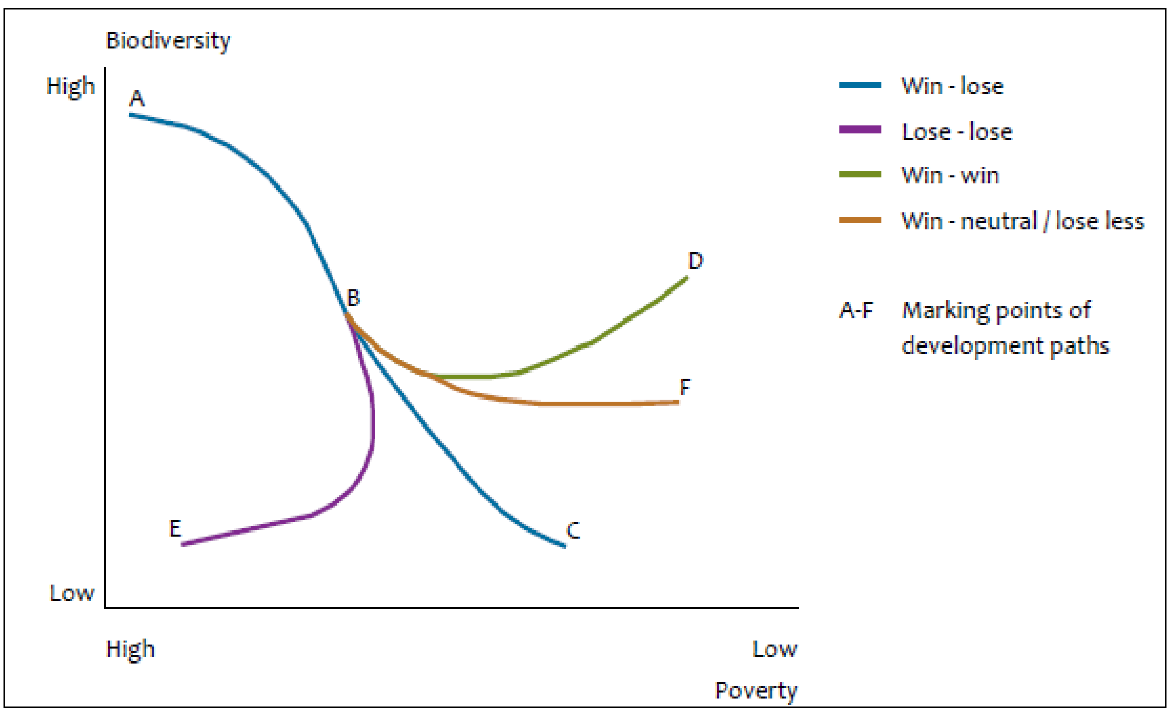


The MEA and TEEB have shown that changes in biodiversity and ecosystem services are affecting human wellbeing in both positive and negative ways. In some cases, these changes have generated negative impacts on human well-being, with the poor being the worst affected because of they are highly reliant on biodiversity for ecosystem services and livelihood strategies (World Resources Institute, 2005, Millennium Ecosystem Assessment, 2005a).

A key point of clarification regarding the biodiversity-poverty nexus debate is that the relationship between *biodiversity* and *poverty* should not be taken as being the same as the relationship between *biodiversity conservation* and *poverty reduction* (Roe, 2008, Secretariat of the Convention on Biological Diversity, 2010b, Roe et al., 2011). This distinction is further clarified below.

With respect to biodiversity and poverty, four main types of relationships have been described concerning the changes in biodiversity and changes in poverty conditions (Tekelenburg et al., 2009): (1) *Win–lose*, where a decline in poverty is accompanied by a decline in biodiversity; (2) *Lose–lose*, where an increase in poverty is accompanied by a decline in biodiversity; (3) *Win–win*, where a decline in poverty is accompanied by an increase in biodiversity; and (4) *Win more–lose less*, where a decline in poverty is accompanied by biodiversity conservation policies (Figure 2.3).

Figure 2.3 Hypothetical prototype courses of change in biodiversity and poverty
Source; Tekelenburg *et al*. (2009)



Concerning these four relationships, the authors arrived at the following six conclusions;

- There are different combinations of trends in changes in poverty and biodiversity
- Biodiversity, poverty, ecosystem services and sustainable development are poorly defined concepts, and are often not quantified
- Different indicators, analytical frameworks, temporal and spatial scales and actor definitions are in use
- The relationship between biodiversity and poverty is a multi-domain, multi-scale and multi-actor issue
- The quantitative relationship between biodiversity and poverty under different conditions is hardly known
- An overarching and convincing theory is lacking.

With respect to biodiversity conservation and poverty reduction, the literature also suggests diverse forms of relationships (Adams et al., 2004, Agrawal and Redford, 2006, Roe et al., 2010). Adams et al. (2004) proposed a typology of four relationships as follows: (1) Poverty reduction and biodiversity conservation are separate policy realms; (2) Poverty reduction is a constraint on biodiversity conservation; (3) Biodiversity conservation should not compromise poverty reduction; and (4) Poverty reduction depends on biodiversity conservation. These policy categories do not preclude the necessity of both poverty alleviation and conservation, but rather express different viewpoints or prioritization of these complex systems.

Furthermore, the relationship between biodiversity and poverty can lead to both negative and positive outcomes. This relationship can be positive when outcomes of biodiversity conservation activities indirectly or directly lead to poverty reduction at local or national levels, or when outcomes of poverty reduction contribute to biodiversity conservation. On the contrary, it can be negative when conservation outcomes directly or indirectly lead to creation or increase in poverty, or when outcomes of poverty reduction mechanisms contribute to biodiversity loss (Roe et al., 2010).

Given the complex and dynamic nature of interactions of poverty and biodiversity, the nature of the relationships between biodiversity conservation and poverty reduction cannot be generalized, and conclusions can only be made for specific cases under specific circumstances (Barrett et al., 2011, Tekelenburg et al., 2009). The current, albeit limited available evidence concerning *biodiversity conservation* and *poverty reduction* (Secretariat of the Convention on Biological Diversity, 2010b) suggests two main points;

a. The poor depend disproportionately on biodiversity for their subsistence needs, both in terms of income and insurance against risk.
b. Biodiversity conservation can be a route out of poverty under some circumstances.

There are however four important caveats to these conclusions. First, it is often the relatively *low value* or ‘inferior’ goods and services from biodiversity that are most significant to the poorest members of the community. Resources of higher commercial value attract the attention of the more affluent groups, often crowding out the poor in the process. Second, even when biodiversity conservation can be shown to make a contribution to poverty reduction, the scale of impact may be limited. Third, a focus on cash benefits obscures the real poverty reduction potential of biodiversity conservation because poverty is not simply the result of low income but also reflects a deprivation of requirements to meet basic human needs. Lastly, in the short-term, biomass may matter more than biodiversity but biodiversity matters in the longer term, particularly as an insurance or risk management strategy for the poor (Secretariat of the Convention on Biological Diversity, 2010b).

In the ASALs, a critical part of biodiversity is the grasslands which provide pasture for domestic and wildlife herbivores, including livestock and large mammals respectively. In Africa, the changes in the conditions of grassland pastures that occur due to either natural factors such as fires or driven by anthropogenic activities such as overharvesting does have significant implications on poverty among pastoral and agro-pastoral communities (CESPA, 2008). There is evidence for example, of significant dependence of poor people on wildlife for livelihood and food security and where wildlife populations are declining or access to wildlife is denied, poor people can cope with these changes, but often at a cost to their livelihoods in terms of reduced income, fewer livelihood diversification opportunities and increased vulnerability (DFID, 2002).

In addition, changes in environmental conditions have implications on the poverty situation of communities living in rural areas that are highly dependent on agriculture and natural resources (World Resources Institute, 2005). The changes in natural resource goods and services as a result of climate change for example, is likely to have a significant impact on the low income poor households through effects on agricultural production processes and changes in household reliance on environmental goods and services (Hertel and Rosch, 2010). In the ASALs, environmental shocks such as severe droughts and floods can erode livestock assets of pastoral households, leading to poverty and increasing their vulnerability to climatic changes (Thornton et al., 2006, Western and Manzolillo Nightingale, 2003).

Although the ASAL areas are prone to repetitive droughts, pastoral and agro-pastoral families may not have sufficient resources to help them mitigate the effects of the shocks that arise from unpredictable droughts. One of the ways in which pastoral families are adapting to the climate variability is through diversification of land use and of their income portfolio into multi-stranded income sources. Specifically, diversification into less drought prone sources of income may increase their resilience by reducing the variability of the otherwise highly volatile income derived from livestock. And because poverty is linked to the vulnerability to climate change in all the three dimensions (i.e. risks to human life and activities, *adaptive capacity* and the processes generating vulnerability), it is critical that poverty reduction interventions in pastoral areas are also tailored towards the reduction of vulnerability of the poor to climate change (Eriksen and O'Brien, 2007).

Since PES income provides pastoral households with a predictable income, including in the dry seasons or drought periods, at least in the short term, it is hypothesized that the diversification of pastoral income and land use to incorporate PES may contribute to climate change adaptation by increasing pastoralist's resilience in the short to medium term (UNCCD/UNDP/UNEP, 2009). Furthermore, because pastoral communities are highly dependent on natural resources, PES can in addition to poverty reduction potentially further serve as a mechanism for promoting ecosystem-based adaptation (Wertz-Kanounnikoff et al., 2011), defined as "the use of biodiversity and ecosystem services as part of an overall adaptation strategy to climate change" (Secretariat of the Convention on Biological Diversity, 2009).

Payments for Ecosystem Services

The definition of PES

Payment for Ecosystem Services (PES) is currently considered as a promising tool for biodiversity conservation and ecosystem management in the developing world (TEEB, 2009). There is no single definition of what constitutes instruments commonly referred to as PES. The most widely used definition defines PES as:

> "(1) a voluntary transaction in which (2) a well defined environmental service (or land use likely to generate that service) (3) is "bought" by a (minimum of one) buyer (4) from a (minimum of one) provider (5) if and only if the provider continuously secures the provision of the service (conditionality)" (Wunder, 2005).

The above definition while theoretically appealing is in practice viewed as too narrow because very few PES programs can meet all the five listed criteria (Engel et al., 2008, Muradian et al., 2010). The PES initiatives that do not meet all the five criteria are termed "PES-like" rather than pure PES programs (Wunder et al., 2008). It is recognized that the scope of PES is wide, with many variations in structure in terms of the form of incentives or payment, the nature of recipients (whether individuals or community), the type of ecosystem services provided, the stakeholders involved (buyers and intermediaries), the rules of participation, and the source of funding (Jack et al., 2008).

The definition of PES suggested by Wunder has been criticized on two grounds. First, it is viewed as being based on an "environmental economics" conceptualization, which promotes the logic that PES should integrate ecosystem services into markets (Muradian et al., 2010). Secondly, it is seen as giving priority to economic efficiency over poverty alleviation concerns (Farley and Costanza, 2010, Pascual et al., 2010, Muradian et al., 2010).

Other alternative definitions of PES have emerged to address these criticisms. Muradian et al. define PES as follows;

> "…a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources" (Muradian et al., 2010, p.1205).

According to these authors, such transfers (monetary or non-monetary) are embedded in social relations, values and perceptions, which are decisive in conditioning PES design and outcomes. Not all PES are therefore based on market transactions. Given the large diversity of PES initiatives, Muradian et al. propose to cluster the PES schemes according to three criteria: (1) the importance of the economic incentive; (2) the directness of the transfer; and (3) the degree of commodification of environmental services.

Another alternative definition of PES is provided by Swallow et al (2009) based on the concept of "Compensation and Rewards for Ecosystem Services (CRES)". The CRES is defined as;

> 'contractual arrangements and negotiated agreements among ecosystem stewards, environmental service beneficiaries and/or intermediaries for the purposes of enhancing, maintaining, re-allocating or offsetting damage to environmental services'(Swallow et al., 2009)

Thus CRES refers to a range of mechanisms linking ecosystem stewards and environmental service beneficiaries and includes payments for ecosystem services. These authors argue that this broad conceptualization is necessary because the relationships between ecosystem stewards, ecosystem service beneficiaries, and intermediaries may be more complex than a simple transaction, with agreements that are not wholly voluntary and payments that are not wholly conditional. Furthermore, they argue, that it is critical to distinguish payments given to ecosystem stewards as a compensation for the damages that they incur and that provided as a reward for good environmental stewardship. The debate over the definition and meaning of PES is an ongoing one, represented by different points of view, ideological orientations and the nature of interest of the stakeholders (Shelley, 2011).

This thesis looks at PES from the perspective of direct payments for biodiversity (Ferraro and Kiss, 2002). Direct payments for biodiversity refer to "*contracts whereby custodians for natural resources are rewarded for biodiversity protection*" and are considered as a sub-set of payments for environmental services (PES) which meets the following three criteria (Milne and Niesten, 2009):

a. Payments are made explicitly for biodiversity conservation, rather than for other environmental services such as carbon sequestration or watershed maintenance (although it may include cases where ecosystem services are bundled).
b. Payments are at least partially funded by philanthropic global investors or international donors and do not rely on local users of environmental services or public funds.
c. Payments are for the protection of existing biodiversity, not for restoration.

Direct payment for biodiversity PES schemes in the context of this study refers to initiatives that involve the following;

a. Contracts between pastoral landholders and government and non-governmental conservation organisations or commercial tourism companies;
b. Explicit payments to landowners for maintaining a stipulated land use that supports wildlife conservation, sometimes jointly with nature-based tourism;
c. Payments in cash made directly to households and not provided through communal institutions.

Thus, a fundamental difference between the definition adopted in this study and that of Milne and Niesten (2009) is how payments are funded; In addition to philanthropic global investor and international donors, I also consider local ecosystem service users, domestic and global corporations in the private sector, and national and local governments among the current and potential funders of direct payment for biodiversity PES program.

Categorization of PES

Most operational PES programs involve payments for four kinds of ecosystem services (Wunder et al., 2008): payments for carbon sequestration especially in forest ecosystems (Jindal et al., 2008); payments for watershed services (PWS) (Porras et al., 2008); payments for landscape beauty; and payments for biodiversity conservation (OECD, 2010). In some cases, payments are made for 'bundled services' that involve selling more than one service at a time (Wendland et al., 2010).

This thesis focuses on payments for biodiversity that directly target wildlife resources and their habitats and may also involve landscape beauty in savannah rangelands. Thus, PES involves a variety of aspects, and therefore can be broadly categorized based on several factors, which include the following;

.

a) *Demand for the ecosystem service*: The demand for ecosystem services originates from three main sectors; the government, private firms, and Non-Governmental Organizations (NGOs) (Lipper et al., 2009). The PES programs in which the payments are provided by the private firms are referred to as 'user-financed' schemes because here the buyers are the actual service users. On the other hand, PES programs in which payments are provided from government and public sources are referred to as 'government-financed' PES schemes because here, the buyers are a third party acting on behalf of the users (Engel et al., 2008). Due to the limited financial capacity of developing countries' governments, money to support publicly funded PES programs are usually sourced from bilateral and multilateral development agencies (Ravnborg et al., 2007). These includes the

Global Environmental Facility (GEF) which provide time limited funds to projects that generate global environmental goods in the area of climate change mitigation, biodiversity conservation, land degradation and the international waters (GEF, 2010).

b) *Type of ecosystem and land use*: PES can be implemented in different natural ecosystems involving different land uses, and within agricultural landscapes (FAO, 2007). The majority of the PES programs that target watershed and carbon services are operational in forest ecosystems. Unlike the forest ecosystem, very few PES schemes have been set up in dryland ecosystems. Some of the ongoing dryland PES programs are being implemented in the developed countries such as Australia (Greiner et al., 2009) and the United States (Goldstein et al., 2011). Only a handful of PES programs are to be found in rangelands in the developing countries (Duttilly-Diane et al., 2007, Victurine and Curtin, 2010). There are virtually no established PES programs in the African rangelands where extensive pastoralism dominates as the main form of land use (Dougill et al., 2012). This means that so far, there is very little experience with the involvement and participation of pastoral and agro-pastoral communities in PES programs (Silvestri et al., 2012).

c) *Labor implications*: Two distinctions can be made regarding the labor requirements and implications of PES activities and regulations for land use (Zilberman et al., 2008). In the first case people can be paid to conserve pre-existing ecosystem services, the so called '*use-restricting*' or '*land-diversion*' PES programs, which involve minimal labor demands. In a second case, people can be paid for land restoration activities, the so called '*asset-building*' or '*working-lands*' PES programs, which are labor demanding. Consequently the latter type of PES programs are likely to have better distributional effects compared to the former (Zilberman et al., 2008) even though both will have different implications for local economic activity, employment opportunities, and thus also for poverty (Wunder et al., 2008).

d) *Property rights and land tenure*: PES can be classified based on the nature of prevailing property rights and land tenure regime because their implementation occurs in one of the four property regimes (Swallow and Meinzen-Dick, 2009), taken here to mean the structure of rights and duties characterizing the relationship of individuals to one another with respect to that particular environmental resource (Bromley, 1991). These regimes are (1) *state property*, where land rights are held by a government that has a right to determine, regulate or subsidize use; (2) *private property* where land rights are held by individuals or firms who can exclude others, but have a duty to refrain from socially unacceptable uses; (3) *common or group property*, where resource rights are held by a group of users or 'co-owners' who can exclude others; and (4) *non-property (or open access)* that is characterized by an absence or breakdown of enforced property rights with no defined group of users or "owners".

e) *Scale of implementation*: PES can be implemented across three different geographic scales. The first is at the *local scale*, for example through upstream-downstream payment arrangements for watershed services. The majority of PES programs that are currently operational are at the local scale (Wunder et al., 2008, Porras et al., 2008). The second is at the *national scale*. While national PES programmes are common in the developed countries, only a few countries in the developing world most of which are countries in economic transition have national PES programmes. These include Costa Rica (Pagiola, 2008), Mexico (Munoz-Pina et al., 2008), and China (Bennett, 2008). In Africa, only South Africa has a national PES programme (Turpie et al., 2008). The third is at the *international scale*, and here a prominent example is the United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries (UN-REDD) for carbon mitigation (Barbier, 2011).

Contrasting perspectives of different interest groups in PES

The implementation of PES involves different interest groups with contrasting perspectives on the role of PES. It is necessary to recognize these diverse perspectives in the analysis of PES and poverty. Table 2.2 presents some of the contrasting perspectives of seven different groups and their perceived outlook of PES whether positive or negative. Clearly all these groups have many positive perspectives on PES. However, from the perspective of farmers and ecosystem stewards (land managers) and poverty reduction, there is on the negative side, a legitimate fear of the use of PES to alienate communal land, especially in rural areas that lack well defined land tenure rights. Indeed, PES has become characterized with the phenomenon of “green grab”(Fairhead et al., 2012).

From the wildlife conservation perspective, the PES mechanism is viewed in three ways. First, it is considered as a new source of finance for conservation, mainly from private sector. Second, it is considered as a tool for the conservation of critical wildlife habitat on private lands (Kiss, 2004b, Nelson et al., 2010, Pagiola, 2003). Lastly, PES is also considered as a tool for mitigating human-wildlife conflicts (Ura et al., 2009). This is especially critical to promote human-carnivore co-existence in ASALs where pastoralists suffer high rates of predation from large carnivores (Dickman et al., 2011, Nelson, 2009, Zabel and Engel, 2010).

Table 2.2 The contrasting perspectives on PES by different stakeholder groups.
Source; Modified from Swallow *et al.* (2009).

Sector	**Perspective**	
	Positive	**Negative**
Farmers and ecosystem stewards	o Official recognition of rights to reside in, use, and modify a protected ecosystem o A new government program that provides public services in exchange for formation of groups or planting trees o A new source of revenue for performing a defined service	o A new way for governments and powerful interest groups to dispossess people from their land.
Wildlife Conservation	o New source of conservation finance o A compensation mechanism for livestock predation and damage to crops o A reward mechanism for provision of	o Doubts regarding PES potential for wildlife conservation

	wildlife habitat	
Poverty reduction	o Alternative income stream for the poor farmers	o Fear of land dispossession from poor people
Rural empowerment and social justice	o A mechanism to redress historical imbalances in the power, rights, and responsibilities of resource-dependent people vis-à-vis ecosystem-service beneficiaries who enjoy greater influence over the political and economic processes	
Environmental management	o A mechanism for the provision of positive incentive for good environmental stewardship to complement regulations o A tool to resolve conflicts over resources access o A mechanism for benefit sharing	
Economic planning	o Flexible mechanism to correct market failure and address collective action challenges	
Business	o Redressing environmental damage caused by business operations as a legal or ethical imperative o A component of a corporate social responsibility strategy designed to maintain or enhance the reputation of the business o Compliance with current or likely future environmental regulations o Sustaining or improving crucial ecosystem services that are inputs into business operations	

The impact of PES on poverty

There is considerable interest on the relationship between PES and poverty particularly in the developing countries where poverty reduction is a major priority (Lipper et al., 2009). Recent estimates suggest that PES could benefit millions of low income households by the year 2030, providing a non-negligible contribution to poverty alleviation at the global level (Table 2.3). Indeed, one of the main reasons driving the rapid growth in PES in developing countries is the view that PES can be a means to reduce rural poverty and support economic development especially through employment creation, income generation, and diversification of livelihoods among ecosystem service suppliers (FAO, 2007, Lipper et al., 2009, Ravnborg et al., 2007).

Table 2.3 Estimates of PES benefits to low income households in developing countries by the year 2030.
Source; (Milder et al., 2010).

Type of PES scheme by ecosystem service	**Number of low income households**
Markets for biodiversity conservation	10–15 million
Carbon markets	25–50 million
Markets for watershed protection	80–100 million
Markets for landscape beauty and recreation	5–8 million

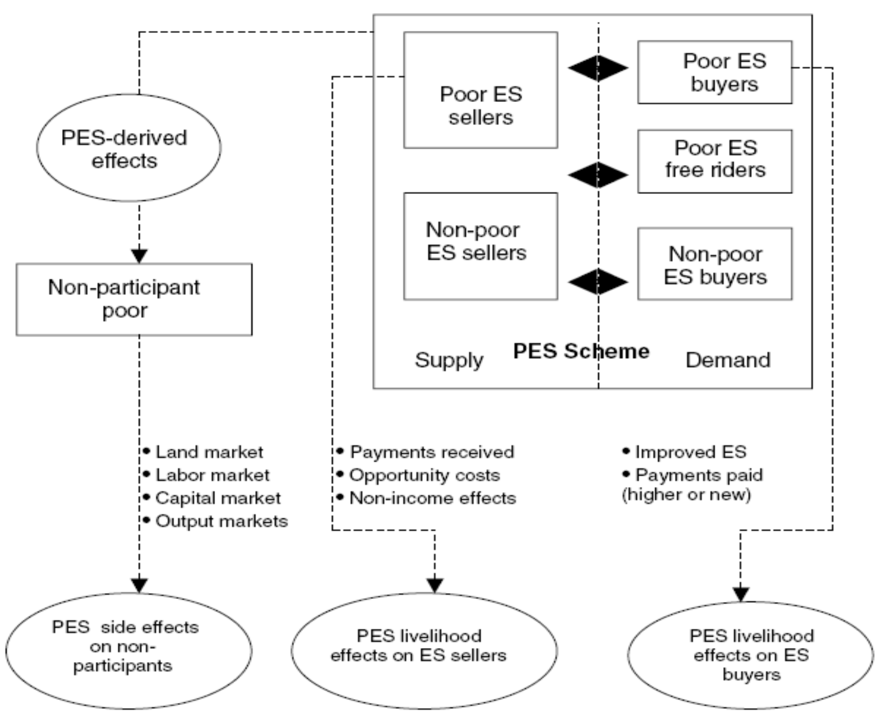
According to Pagiola et al. (2005), there are two main questions regarding the potential impact of PES on poverty reduction. The first is whether PES can reduce poverty among participating households and, indirectly, nonparticipants in target areas. The second is the broader macro-economy question of whether PES can contribute towards the reduction in poverty at the national level (Pagiola et al., 2005). Both of these questions are addressed in this thesis, except that the second question is addressed mainly with specific reference to the Kenyan ASAL regions.

An initial review of the research on PES impacts on poverty highlighted a major gap, which directly informed the focus of this study. To date, the evidence of PES impact on poverty is based on payment for watershed services (PWS) and payments for carbon sequestration PES programs. The empirical studies of PES programs targeting biodiversity conservation (including wildlife) are lacking. This dearth of evidence is acute for pastoral rangelands in Africa that are rich in wildlife resources but beset by high poverty levels. Consequently there is little understanding of the involvement and potential poverty impact of PES among pastoral and semi-nomadic communities that are among the most economically disadvantaged groups (Little et al., 2008, Homewood, 2004).

This study takes as its point of reference, the conceptual framework of the impact of PES on the poor presented in Figure 2.4. This framework is applied in the review of the literature and in the analysis of the PES programs presented in the chapters 4, 5 and 6. In this conceptual framework, it is noted that a typical PES scheme consists of the demand side with buyers and the supply side with the sellers of Ecosystem Services (ES) respectively. Theoretically, there are poor and non-poor ES sellers and ES buyers as well as non-participants that are affected by PES. Thus the implementation of PES will generate livelihood effects on the ES buyers, the ES sellers and non-participants in different ways.

The poverty effect on the demand side may occur through the improvement in ecosystem services and payments provided by ecosystem buyers. On the supply side, the PES components that does affect poverty include the payments received, the opportunity costs and other non-income effects. Land, labor capital and output markets are the livelihood components of non-participants affected by a PES scheme. The main focus of this study is on the supply side of a PES scheme and concerns both the poor and non-poor ES sellers. The study therefore does not look at poverty on the demand side of a PES scheme.

Figure 2.4 A conceptual framework on the impact of PES scheme on the poor.
Source; (Wunder, 2008).



A key shortcoming of this conceptual framework is the exclusion of PES Intermediaries. The analysis in this thesis addresses this gap by considering and assessing the role of PES intermediaries in the two case studies reviewed. The links between PES and poverty follows four sequential questions (Wunder, 2008). The first is to what extent poor people participate in PES schemes as buyers and sellers of environmental service (so called "participation filter"). The second is, if the poor become service sellers, does this make them better off (the "effect on sellers"). Thirdly, do poor service buyers (and non-paying poor service users) become better off from PES (the "effect on users"), and lastly, how are other, non-participant poor affected by PES outcomes (the "derived effect"). A review of the current state of evidence in respect of these four questions is presented below.

To what extent do poor people participate in PES schemes? (Participation filters)
The direct benefits from PES in terms of cash payments typically accrue to households that are eligible, willing and able to participate in PES schemes. Thus the rules for eligibility and household ability to participate in a PES scheme are critical as these can limit the participation of poor households (Pagiola et al., 2005). At least seven factors have been noted to determine the participation of poor households and the pro-poor outcomes of PES schemes. These include the eligibility criteria determining who has access to participate as potential ecosystem service providers; the type of ecosystem service and paid-for management practice; the institutional options for dealing with transaction costs, including co-operative institutions and bargaining power among poor providers; the type and level of payment; the level of legal and institutional equity; the level of awareness, education and technical capacity; and the nature of property rights (Pagiola et al., 2005, Wunder, 2008, Ravnborg et al., 2007, van Noordwijk et al., 2012).

Currently, the evidence regarding the participation of the poor in PES schemes is mixed. On the one hand, several studies have shown that poorer households are equally able to participate in PES programs as much as their non-poorer counterparts (Pagiola et al., 2007, Pagiola et al., 2008, Pagiola et al., 2010). On the other hand, cases have been documented where the participation of the poor households in PES have been limited by the high transaction costs involved, institutional and technical barriers, lack of information, and weak capacity for negotiation (Iftikhar et al., 2007, Kosoy et al., 2008, Zbinden and Lee, 2005). The high transaction cost in particular is a major obstacle to the participation of the poor in virtually all PES programs evaluated for their poverty impacts (Grieg-Gran et al., 2005, Pagiola, 2007, Pagiola et al., 2008). In addition, the poor have also been disadvantaged under regimes of communal property, such as in Mexico, where better-off *ejidos* were over-represented in the national PES programme compared to the very highly marginalized *ejidos* that were substantially under-represented (Munoz-Pina et al., 2008).

Does participation in PES make the poor sellers better off? (Effects on sellers)

Among households already enrolled in a PES program, the key poverty issue is to what extent poor households can and do benefit from program participation. It is recognized that PES can benefit the poor participants directly through the provision of cash flow, as a fungible store of wealth, and as a means of promoting household income diversification, and indirectly through social and cultural gains (Grieg-Gran et al., 2005, Lipper et al., 2009, Landell-Mills, 2002). Yet, a recent review of evidence concerning the effectiveness of biodiversity conservation mechanisms, including PES concluded thus;

> "Overall, PES is rarely a route out of poverty but does have limited poverty reduction benefits. When PES is not truly voluntary, it can become a poverty trap, though this has rarely been documented" (Secretariat of the Convention on Biological Diversity, 2010b).

The evidence generated from a meta-analysis of PES schemes shows that PES can contribute to positive, albeit mostly marginal income gains for poor families in these programs (Corbera et al., 2009, Wunder, 2008). In some cases, the research shows that cash payments can be significant relative to household income/expenditure. In Pimampiro, Ecuador for example, the cash income derived by household from PES were found to correspond to 31% of total household expenditure (Wunder and Alban, 2008). This is however an exception, as in many other cases, the cash income from PES was found to be relatively unimportant, often less than the estimated financial opportunity costs of participating households (Corbera et al., 2007b, Kosoy et al., 2007).

Other examples where PES has provided only marginal contribution to the share of household income have been documented in a PES scheme in Jesus de Otoro, Honduras where the PES payments provided was found to be less than 1.2 % of the participating household gross income and in two other PES programs in San Pedro del Norte, Nicaragua where PES payments comprised less than 10% and 1.2% respectively of the annual household income of participating households (Corbera et al., 2007b, Kosoy et al., 2007).

Apart from the cash income, the households participating in PES can also gain from non-income benefits. These include benefits through increased land tenure security especially under weakly defined property rights, and also the strengthening of social institutions (Grieg-Gran et al., 2005, Greiner and Stanley, 2013, Engel and Palmer, 2008). In Mexico, participating households reportedly benefited from complementary PES project activities such as forest management and agricultural training support (Corbera et al., 2007b, Corbera et al., 2009).

Do poor service buyers become better off from PES? (Effects on users)

Little research has so far been conducted on the implication of PES on poor ecosystem service buyers and consequently not much is currently known regarding the PES effects on poor users. The majority of the existing PES schemes lack direct poverty alleviation scope on the buyer side because it is frequently assumed that potential ecosystem service sellers are poorer than buyers/users, hence PES related poverty alleviation initiatives are focused on the former (Wunder, 2008).

How are other, non-participant poor affected by PES outcomes? (Derived effects)

While PES effects are likely to be neutral for most non-participating poor people, some may be affected significantly through changes in land, labor, capital and output markets (Wunder, 2008, Zilberman et al., 2008). Regarding land market effects, it’s noted that

PES can provide incentives for powerful groups to take control of currently marginal land, thereby excluding the poor and disenfranchising the landless that are looking for opportunities to invade land for occupation (Landell-Mills, 2002, Swallow and Meinzen-Dick, 2009, Asquith et al., 2008).

With regard to the labor market effects, “use-restricting” PES schemes can generate risks to the poor non-participants, for instance, through loss of security and control of land and restrictions to resource access, some of which may lead to negative impacts on the landless, the herders and women, by restricting their access to commons such as grazing lands, and to non timber products from forests (Kerr, 2002, Lee and Mahanty, 2009, van Noordwijk et al., 2012).

This review of evidence clearly shows that much remains to be determined with regard to the impact of PES on poverty and this is pertinent for dryland ecosystems. In this study, I will address three of the four questions highlighted above which deal with the supply side of a PES scheme. I however do not address the question “Do poor service buyers (and non-paying poor service users) become better off from PES?” as this is beyond the scope of this study.

Ecosystem and wildlife population changes and poverty in Kenyan ASALs

The field research reported in this thesis was carried out among pastoral communities in southern Kenya’s arid and semi-arid land (ASAL) between 2008 and 2010. This was a few years following the publication in 2005 and 2007 of two reports on poverty and ecosystem services in Kenya (Duraiappah and Marlene, 2007, Wong et al., 2005), and a Kenyan Atlas of ecosystems and human well-being (World Resources Institute, 2007). These publications documented the deterioration of many ecosystems in Kenya especially in the ASALs leading to stress on a range of ecosystem services. The deterioration in ecosystem services in turn contributes to the declines in human well-being and to increased poverty as a result.

In tandem with reported ecosystem degradation, and declines in human wellbeing, ecologists reported high losses in biodiversity in the Kenyan ASALs. This was based on the assessment of changes in wildlife populations that were found to be decreasing at an alarming rate (Western et al., 2006). On average, the population of large herbivores, more than 75% of which are found in ASALs, declined by 3.2% annually between mid-1970s and mid-1990s (Norton-Griffiths, 2007a). These estimates suggests that on aggregate, wildlife populations in Kenyan parks and the adjoining areas declined by 48% and 45% respectively in the 20 year period between 1977 and 1997 (Western et al., 2009).

Currently, it is estimated that Kenya's national parks now contain only 10% of the large mammal wildlife populations with the remaining 90% dispersed in the Maasai Mara National Reserve (25% of population), private and communal Conservancies (40% of the population) and in non-protected public and communal lands across much of the ASAL areas (25% of the population) (Table 2.4). Wildlife populations are declining everywhere with the exception of private and communal Conservancies. These have recorded an increasing trend in wildlife populations suggesting that they are probably more effective than state protected areas in terms of wildlife conservation (Western et al., 2006).

Table 2.4 The distribution of wildlife in Kenya by conservation land category and direction of trend (positive or negative) in bracket.
Data source; (Western et al. 2009)

Conservation land category	Share of population of wildlife in Kenya (%)	Trend in wildlife population
Private and communal wildlife conservancies	40	Positive (population increasing)
Maasai Mara National Reserve (Protected area)	25	Negative (population decreasing)
Non protected public and communal lands	25	Negative (population decreasing)
Public protected areas	10	Negative (population decreasing)

There could be many reasons to explain the high declines in wildlife population in Kenyan ASALs in the period from 1970s to 2000s. Apart from poaching, and the possible effects of periodic droughts, these massive wildlife losses are principally driven by land use changes especially the expansion of agriculture in marginal rangelands (Norton-Griffiths and Said, 2010). The conversion of rangelands to crop cultivation leads to habitat loss and fragmentation, especially affecting wildlife migratory corridors and dispersal zones outside protected areas (Galvin et al., 2008). Changes in property rights through land privatization also accelerate the conversion of rangelands to cropland (Homewood et al., 2009c). When land is privatized and sub-divided, the new pastoral landowners especially in the wetter and the high potential rangelands lease out land for large scale commercial crop cultivation (Thompson et al., 2009).

Concurrent with the land use changes and declining wildlife populations in ASAL, is the high income poverty rates and the marginalization of pastoral communities in these regions. Analysis of poverty show that in terms of income and non-income indicators (including education, health and nutritional levels), the majority of the pastoral communities in Kenya are poor hence the relatively high poverty prevalence recorded in the ASALs (Okwi et al., 2007, Watkins and Alemayehu, 2012). Paradoxically, the high poverty rates in the ASALs prevail against a backdrop of high value resources. For example, with an estimated 75% of Kenya’s wildlife resources and 88% of wildlife protected areas, the ASAL serve as the backbone of Kenya’s thriving wildlife dependent tourism industry which generates millions of dollars in revenue and consistently ranks among the top contributors to the national GDP (World Bank, 2011).

The lack of a transparent benefit-sharing mechanism in the Kenyan wildlife tourism sector means that a large share of the tourism revenue is diverted away from the landowners that host wildlife. It is estimated that landowners, including pastoral communities are able to capture only about 5% of total wildlife tourism revenue yet landowners bear the largest share of wildlife related risks and costs in the form of competition for forage and pasture, water and space, livestock predation by carnivores, and human-wildlife conflicts, including injury and loss of life (Norton-Griffiths, 2007a, Sindiga, 1995). Consequently, many poor pastoral landowners have very little or no incentive to retain wildlife on their private land especially in the absence of direct wildlife benefits in the form of tourism revenues or other wildlife enterprises (Norton-Griffiths, 1998, Kameri-Mbote, 2002, Kabiri, 2010).

With the evidence of increasing wildlife populations in Conservancies located on private and communal lands, in contrast to the state protected areas and non protected public and communal lands which have recorded declines in wildlife populations (see Table 2.4), conservation practitioners are now advocating for the expansion of community conservancies in ASALs, and the use of market based instruments including PES as one of the mechanisms to halt the recorded declines in wildlife populations through the loss of wildlife habitat on private and communal lands, and to guarantee direct economic benefits for pastoral landowners (Norton-Griffiths, 2007a). Thus the PES programs that are emerging in Kenyan ASAL areas focused concurrently on the dual goals of wildlife habitat conservation and poverty reduction among pastoral landowners (Bulte et al., 2008a), but with very little experience on the ground, it remains to be determined to what extent the existing PES programs are able to realize these dual goals.

Selection of Study Sites and PES programs

This chapter section describes the selection of study sites and PES program case studies. An inventory of PES in Kenya, Uganda, Tanzania and South Africa in 2006 and 2008 by the *Katoomba Group* showed a high uptake rate, with the number of PES schemes increasing by 64% (from 45 to 74 schemes) in the four countries, and by 54% (from 17 to 26 schemes) in Kenya (Bond et al., 2008). No updated data on the number of PES in the region were available for 2012.

To begin to establish the evidence on the impact of PES on poverty among pastoralists in Kenyan ASALs, I first conducted a scoping assessment of existing PES schemes in Kenya, as part of this study. This was accomplished through an evaluation and review of peer-reviewed and grey literature, extensive internet search on the worldwide web, and through contacts with key informants drawn from government and non-governmental institutions that are active in PES in East Africa: The World Bank; the International Livestock Research Institute (ILRI); the World Agro forestry Center (ICRAF); National Environment Management Authority (NEMA) in Kenya; the World Conservation Union (IUCN); and the World Wildlife Fund (WWF).

Following the review, I developed six criteria against which to screen candidate PES projects that have one or both objectives of wildlife conservation and poverty reduction (Table 2.5). Given the limited experience with PES in the Kenyan ASALs, the purpose of the screening was not to identify PES projects that met all the six shortlisted criteria but rather to select those with sufficient empirical data on both the poverty and ecosystem services dimensions.

Table 2.5 A shortlist of criteria for screening candidate PES programs and study sites

No.	Criteria	Notes
	PES scheme design aspects	
1	PES scheme based on direct payment model	Consider a scheme if it provides cash and not in-kind payments and this is remitted directly to household
2	longevity	Consider a scheme that has been implemented for not less than two years with available data on enrolment and payments
	Ecological factors considered in the selection of the sites of PES intervention	
3	Biodiversity importance of site/ecosystem	Consider a scheme if it is located in an area of ecological or conservation importance such as near a protected area or has significant populations of wildlife
4	Threats to biodiversity (ecosystem and wildlife)	Consider a scheme if it is operational within an ecosystem facing high threats as defined either by (1) “critically endangered status” or “endangered status” in the Draft National Wildlife Bill (2007 and 2009), or (2) other conservation listing criteria (e.g. WWF Ecoregion, BirdLife’s Important Bird Area etc)
		Consider a scheme if it is located within an ecosystem with established high declines in populations of wildlife species especially of migrant herbivores such as wildebeest and zebra
	Socio-economic factors considered in the selection of the sites of PES intervention	
5	Poverty levels	Consider a scheme if it is located in an area with high poverty levels based on government poverty statistics (e.g. the KNBS surveys) or local studies
6	Tourism visitation and revenues	Consider a scheme if it is located within a site with relatively high wildlife revenue generated through tourism or other wildlife activities (minimal)

The process of case study selection also involved a national level survey of the emerging community based wildlife conservation initiatives that includes Conservancies and wildlife PES schemes located on group and individually owned pastoral lands in Kenyan ASALs. The findings of the survey are presented in Chapter 3. I selected two PES candidate programs in southern Kenya that met all the six screening criteria. The following three considerations were especially critical in the study sites and the PES programs selected;

- The sites are currently classified as "*Critically Endangered*," suggesting that some ecosystem services are already severely degraded (Government of Kenya, 2007).
- Despite being located adjacent to protected areas with high tourist visitation and revenue generation, the two sites have high poverty prevalence.
- The pilot PES programs have already been operational for more than two years. These PES programs involve direct payments to pastoral landowners contingent on their maintaining stipulated wildlife-friendly land use on enrolled land parcels.

The PES schemes selected are the Wildlife Lease Program (WLP) in the Athi-Kaputie Plains (AKP) and the Olare Orok Conservancy (OOC) in the Maasai Mara Ecosystem (MME). The programs involve pastoral Maasai land owners living on critical wildlife dispersal and migration routes adjacent to the wildlife protected areas of Nairobi National Park (NNP) and the Maasai Mara National Reserve (MMNR) respectively (Figure 2.5).

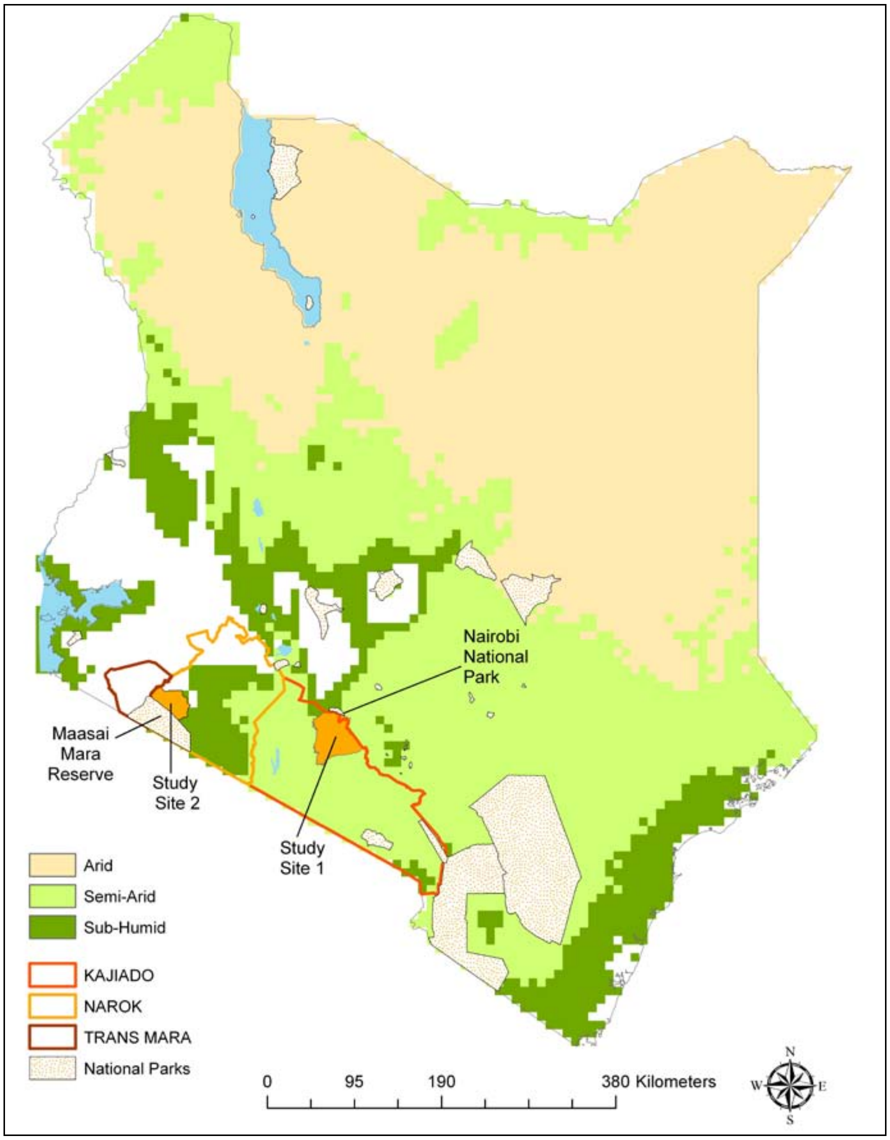
The two PES schemes assessed here were established to address two primary concerns. The first is the regulation of habitat loss and fragmentation arising from the expansion of crop cultivation, fencing and land subdivision that constraint both the dispersal and migration of wildlife in and out of protected areas, diminishing the potential for tourism and constraining the seasonal mobility of pastoralists with their livestock.

The second is the need to enable pastoral landowners to derive direct financial income for wildlife protection, including from the state wildlife management agency (the Kenya Wildlife Service: KWS), and from the commercial tourism companies that are investing in tourism operations in the wildlife dispersal areas. These direct benefits to pastoral landowners from tourism have hitherto been lacking, especially in around the Maasai Mara National Reserve. A critical factor in both sites is the change in land tenure, from communal land ownership to private, individual or corporate ownership.

The two PES programs share many characteristics as shown in Table 2.6, but there are also some fundamental contrasts that exists regarding the location and spatial distribution of landowners enrolled, which is as a result of the differences in how these PES programs were set up. In the Wildlife Lease Program (WLP) in Athi- Kaputie Plains, land owners first received their land parcels and title deeds following land privatization and sub-division which took place in the area in late 80s to early 1990s (Rutten, 1992). By the time the WLP was started as a pilot PES program in the year 2000, land in the AKP had already been fully privatized and individuated. Consequently, land owners were then progressively recruited in the WLP based on an understanding that they could be able to withdraw from the program if they wished. Because landowners in different parts of the AKP enrolled separately, the WLP ended up in a checker-board pattern with participants dispersed widely across the landscape in AKP.

In contrast, in the Olare Orok Conservancy (OOC) the area targeted for creation of the Conservancy was delimited at the same time as land sub-division and issuance of title deeds was ongoing from 1999 to 2004. Thus, only households that were allocated land within the zone delimited for the Conservancy which comprised a single block with multiple contiguous land parcels were recruited to be part of the Conservancy. Thus because landowners were restricted to a single contiguous block of land, the OOC ended up as one large block of land. Consequently, although the households enrolled in the OOC in principle have the option of withdrawing, in practice withdrawal can present a huge challenge of access to parcels which are located inside the Conservancy.

Figure 2.5 Map of Kenya showing the arid, semi-arid and sub-humid areas and the location of the study sites.
Study site 1 is the Athi-Kaputie Plains located to the south of Nairobi National Park. Study site 2 is the Olare Orok Conservancy situated within the Maasai Mara Ecosystem on the northern boundary of the Maasai Mara National Reserve.
Source: International Livestock Research Institute (ILRI).



A description of characteristic of the two study sites and the associated PES schemes are summarized in Table 2.6 below.

Table 2.6 Description of the characteristic of the study sites and the associated case study PES schemes selected for evaluation.

Study site	**Athi-Kaputie Plains**	**Maasai Mara Ecosystem**
Protected area (PA)	Nairobi National Park	Maasai Mara National Reserve
Size of protected area (PA) (ha)	11,700	150,000
National share of wildlife population in Kenya	negligible	25%
National share of Kenya's annual visitors to PAs	22.7%	13.4%
Poverty rate (25Km of PA buffer)	40%	63%
Threat status of ecosystem	Critically Endangered	Critically Endangered
Key threats to ecosystem	Blockage of migration route Land subdivision Urban sprawl	Reducing habitats due to land subdivision Unplanned development Negative environmental impact of tourism
Land Tenure	Private (sub-divided in the late 80s and early 90s)	Private (sub-divided in 1999-2004) and on-going
PES program	**Wildlife Lease Program**	**Olare Orok Conservancy**
Year started (period of operation)	2000 (12 years)	2006 (5 years)
PES payment rate (US$.ha.year^{-1}) in 2010	US$10 (constant)	US$ 43 (variable)
No. of landholders participating in the PES scheme (2010)	357	157
Area of PES scheme (2010)	16,700ha	10,000ha
Contract arrangement (period)	Individual contract (1 year)	Group joint contract (5 & 15 years)
Funding source	Public (Government and Conservation NGOs)	Market (Private tourist companies)
Intermediary	NGO (The Wildlife Foundation)	Private management company (Ol Purkel Ltd)
Conditionality of PES (land use restrictions)	No land sales and sub-division No fencing No crop cultivation on enrolled land	Exclusion of settlements Restrictions on livestock grazing inside the conservancy

Research questions addressed in the thesis

This study addresses three broad questions concerning biodiversity, ecosystem services and poverty in Kenyan ASAL regions. These key questions are as follows;

1. What are the changes and the current status of selected ecosystem services in Kenyan ASAL at the national, regional and local levels? The ecosystem services considered are the provisioning services of livestock production and crop cultivation at the national level (chapter 3); the habitat services at the national (chapter 3), regional (chapter 3) and local levels (chapter 5 and 6); and the recreational services of tourism at the regional level (chapter 3).

2. What are the influences of the direct and indirect drivers of change on biodiversity (using wildlife populations as a proxy), ecosystem services and poverty in Kenyan ASALs? The direct driver considered is climate change as manifested through changes in drought occurrences (Chapter 3 and 4), and the indirect drivers are the demographic changes (chapter 3) and land policy changes affecting land tenure at the national (chapter 3) and local levels (chapter 6).

3. What are the actual and potential effects of PES on poverty at the national (chapter 3) and local levels (chapter 5 and 6)

The subsequent substantive chapters of this thesis respond to these three questions in detail as outlined in Table 2.7.

Table 2.7 The list of substantive thesis chapters and the respective research questions.

Chapter 3: *Evaluating The Potential of Payment for Ecosystem Services for Poverty Alleviation In Relation to Changes In Biodiversity And Ecosystem Services In Arid And Semi-Arid Lands In Kenya*
1) What is the current status of the provision of wildlife habitat services in private lands (Conservancies and wildlife PES schemes) in Kenyan ASALs? 2) How has biodiversity (using wildlife biomass as a proxy), and the supply of ecosystem services changed in Kenyan ASALs in general and Maasailand areas in particular? 3) What effects do the changes in land policies and demography have on biodiversity and the supply of ecosystem services in Kenyan ASALs? 4) What are the potential impact and implication of wildlife PES on poverty and economic inequality in ASAL in general, and Maasailand in particular? 5) How has climatic variability affected pastoral livelihoods in the ASALs especially in terms of drought effects?
Chapter 4: *Exploring The Role of Payments for Environmental Services In Ecosystem Based Adaptation to Climate Change And Poverty Alleviation: Insights from Kenyan Rangelands*
1) What is the trend in drought occurrence in the two study sites based on recorded frequency and severity? 2) What are the effects of PES on the adaptive capacity of pastoral households as environmental service providers? 3) What are the effects of PES on the institutions relevant to climate change adaptation at the local level in the two study sites?
Chapter 5: *Poverty, Inequality and Participation of Pastoralists In A Payment for Ecosystem Service Scheme Adjacent to A Semi-Arid Protected Area In Southern Kenya*
1) What is the nature of the design and implementation of the WLP? 2) What is the level of poverty and inequality among the households in the study area? 3) What are the determinants of participation, and intensity of participation in the WLP? 4) What are the effects of the PES on household poverty, inequality, and income?
Chapter 6: *Why Keep Lions Instead of Livestock? An Assessment of The Effect of Wildlife Tourism- Based Payment for Ecosystem Services on Herders In the Maasai Mara Ecosystem, Kenya*
1) How has land tenure evolved in the Maasai Mara Ecosystem? 2) What is the nature and design of the PES scheme in the Olare Orok Conservancy? 3) What is the level of poverty and wealth inequality among households in the study area? 4) What are the effects of the PES on household poverty, inequality, income and expenditure?

CHAPTER 3: EVALUATING THE POTENTIAL OF PAYMENT FOR ECOSYSTEM SERVICES FOR POVERTY ALLEVIATION IN RELATION TO CHANGES IN BIODIVERSITY AND ECOSYSTEM SERVICES IN ARID AND SEMI-ARID LANDS IN KENYA

Philip M. Osano, Caroline Bosire, Mohammed Said, Shem Kifugo, Jan de Leeuw

Abstract

We explore the potential for wildlife PES to alleviate poverty in the Kenyan Arid and Semi-Arid Lands (ASALs) in relation to changes in wildlife biomass, selected ecosystem services, land use, demography and climatic variability. Our results show an overall decline in wildlife biomass, a corresponding increase in area under crop cultivation, and an overall decrease in livestock production (except for the wetter semi-arid areas) between 1970s and 2000s. The supply of wildlife habitat services in private lands increased substantially between 1992 and 2010, currently covering close to a million hectares or 2% of the total Kenyan ASAL area. In Maasailand, the supply of tourism services increased four-fold between 1999 and 2010, concurrent with the supply of wildlife habitat services in private lands which now exceeds that in state protected areas by some 26%. The demand for tourism only increased marginally by contrast leading to a huge and increasing over-supply of tourism services that risks undermining the long term sustainability of wildlife tourism related PES schemes.

We roughly estimate that 210,000 pastoral people are impacted directly and indirectly by the wildlife conservation initiatives assessed leading to both costs and benefits. The correlation between the occurrence of high poverty rates and high wildlife biomass points to the potential for wildlife PES schemes to impact on poverty. Not accounting for the opportunity costs incurred, an estimated annual PES transfer of US$ 10-15 per hectare would be sufficient to close the poverty gap in much of the ASALs. Given the current limited involvement of the private tourism sector in large areas of the ASALs wildlife PES in areas without viable commercial tourism may only be realized with financial input from the public sector. In terms of climatic variability over the last 40 years, increased temperatures and reduced precipitation have been recorded and both trends are predicted to continue in the short term to 2025.

Introduction

The nexus of poverty reduction and biodiversity conservation is core to environment and development policies (Sachs et al., 2009, Tallis et al., 2008) despite insufficient empirical evidence of the connection between the two (Barrett et al., 2011). The concept of ecosystem services is a common framework that links human well-being, including poverty reduction to biodiversity and ecosystems that support human well-being through provisioning, regulating, habitat, and cultural ecosystem services (TEEB, 2010).

Despite concerted efforts to design projects that combine the objectives of poverty reduction and biodiversity conservation, only in a handful of cases report “win-win” outcomes for both (Tallis et al., 2008, Kareiva et al., 2008, Andam et al., 2010) as many other projects fail to realize one or both objectives (Tallis et al., 2008, Agrawal and Redford, 2006). Nowhere is the challenge of addressing poverty and biodiversity concurrently more critical than in Africa, the world’s poorest continent (Collier, 2007). The need is acute in the arid and semi-arid lands (ASALs) renowned for their richness in wildlife found on land managed by pastoral livestock keepers. Here the perceived degradation of ecosystem services – the expression of a persistent decline in the ability of a dryland ecosystem to provide goods and services associated with primary productivity (Safriel and Adeel, 2005), declines in wildlife population and habitat loss occurs alongside widespread poverty among the pastoral populations (Homewood, 2004, CESPA, 2008).

The ASALs provide a range of ecosystem services, including food and fiber, water provision, climate regulation, and wildlife habitat (Safriel and Adeel, 2005). The most dominant socio-economic activity is the supply of provisioning services, through extensive livestock production of meat, milk and other livestock products. While these provisioning services are marketed, many other ASAL based ecosystem services are not marketed, and remain outside the formal economy, leading to undervaluation and policy neglect of ASALs (Rodriguez, 2008, Barrow and Mogaka, 2007). Consequently, there is little or no rewards or compensation for pastoralists responsible for globally valuable ecosystem services generated through their land stewardship (Davies, 2008).

As an example, although pastoral lands provide critical habitats for wildlife species, including large mammals, which underpins tourism based cultural ecosystem services in East African ASALs, only a paltry proportion of the wildlife tourism revenues currently accrues to pastoral landowners (Norton-Griffiths, 1996). The majority of East African pastoralists live in abject poverty despite occupying areas of rich wildlife and high touristic values (Homewood et al., 2012, Little et al., 2008). There is however considerable potential for wildlife to benefit pastoral communities and contribute to poverty alleviation (DFID, 2002) but this will be difficult to realize in the face of current loss of biodiversity, large declines in wildlife populations and degradation of ecosystem services in ASALs (CESPA, 2008, Western et al., 2009).

The loss of biodiversity and changes in ecosystems have considerable effects on ecosystem services and human well-being in Kenyan ASALs where pastoral livestock grazing and wildlife conservation are the main land uses (Wong et al., 2005, Duraiappah and Marlene, 2007, World Resources Institute, 2007). The ASALs are of critical importance to the Kenyan economy particularly in the agricultural livestock and tourism sectors. The ASALs contain 70% of cattle, 87% of sheep and 91% of all goats in Kenya (Behnke and Muthami, 2011) as well as the majority (>90%) of large wild herbivores, 70% of which live permanently or seasonally outside the wildlife protected areas (Norton-Griffiths and Said, 2010).

The direct and indirect drivers of biodiversity loss have affected the supply of ASAL based ecosystem services. The direct drivers include habitat change, climate change, invasive species, over-exploitation and pollution (Secretariat of the Convention on Biological Diversity, 2010a). Land use changes, particularly the expansion of crop cultivation in rangeland, has contributed to habitat loss and fragmentation affecting pastoral livestock production and wildlife in ASALs (Behnke, 2008) while the high climatic variability especially through drought effects have also increased the vulnerability of pastoral livelihoods to climate change(Birch and Grahn, 2007).

The main indirect drivers include the rapid increase in human population (UNEP, 2009), policy changes involving land tenure, specifically the privatization and sub-division of rangelands (Mwangi and Ostrom, 2009b), and other external drivers especially land grab displacing and dispossessing pastoral ASAL residents(Lambin and Meyfroidt, 2011). These drivers have generated increased pressure on ASAL environmental resources including diminished pastures, reduced water availability, per capita reduction in land (Kabubo-Mariara, 2005) and livestock holdings. These pressures have in addition to biodiversity loss, also created scarcity in the supply of ecosystem services leading to changes in the human benefits derived from ASAL ecosystem services thereby affecting human well-being particularly through increased poverty (CESPA, 2008, World Resources Institute, 2007) as a result of which the ASAL areas are characterized by high poverty rates (Central Bureau of Statistics, 2003, Central Bureau of Statistics, 2005, Okwi et al., 2007).

In response to the loss of biodiversity and increased poverty, some pastoral communities have since 1990s allocated their land to conservation through Conservancies which supply wildlife habitat services on their private and communal lands. The lands and institutions that deliver these wildlife habitat services are commonly referred to as “Conservancies”. Thus, a Conservancy involves the allocation of communal or individual owned land for wildlife conservation and wildlife tourism to generate financial and non-financial benefits directly or indirectly to landowners. These benefits can be provided by commercial tourist companies, conservation NGOs, and the state wildlife and protected area agencies (Sindiga, 1995, Carter et al., 2008).

Conservancies can be distinguished based on a combination of the (i) the conservation approach with a distinction between the “project based” and the “non-project based” models (Kiss, 2004b), and (ii) the directness of incentives (direct or indirect) (Ferraro and Kiss, 2002). Project-based models focus on implementation of a specified set of activities which are expected to result in the preservation of a particular segment of biodiversity, within a stipulated period of time while non-project based models focus on the creation of incentives to reward results as opposed to defining and implementing activities (Kiss, 2004b).

The focus in this chapter is on Conservancies that operate on a “non-project” model, through direct payment for biodiversity (Ferraro and Kiss, 2002). We consider direct payments as version of payments for environmental services (PES) defined as “a voluntary, conditional agreement between at least one ‘seller’ and one ‘buyer’ over a well-defined environmental service – or a land use presumed to produce that service (Wunder, 2007).

In this chapter, we use the term “wildlife-PES” to refer to wildlife conservation initiatives that meet the following three criteria; (1) Involve voluntary contracts between landholders (in group or individually owned lands) and public institutions (government agencies and NGOs) promoting biodiversity conservation or private sector (especially tourist industry); (2) Where payments in cash are made explicitly for a land use that supports wildlife conservation, sometimes also in combination with nature based and cultural tourism; and (3) Where payments are made in cash directly to households and not to community institutions or allocated to communal projects. Thus, wildlife-PES is implemented in some Conservancies but not necessarily in all and there exists wildlife-PES schemes that are operational without necessarily being part of a Conservancy.

In Wildlife PES schemes, payments are supposed to be conditional on conservation results and outcomes (Wunder, 2007, Ferraro and Kiss, 2002). Also, as a voluntary mechanism, it is presumed that landowners will only enroll in PES if it does not make them worse-off in terms of poverty (Pagiola et al., 2005). Despite the current interest, little is known regarding the potential of wildlife PES for poverty reduction among pastoral Maasai (Bulte et al., 2008a). This knowledge gap is considerable especially in relation to the observed changes in biodiversity and ecosystem services.

This study attempts to fill this knowledge gap and specifically respond to the following five questions:

1) What is the current status of the provision of wildlife habitat services in private lands (Conservancies and wildlife PES schemes) in Kenyan ASALs?
2) How has biodiversity (using wildlife biomass as a proxy), and the supply of ecosystem services changed in Kenyan ASALs in general and Maasailand areas in particular?
3) What effects do the changes in land policies and demography have on biodiversity and the supply of ecosystem services in Kenyan ASALs?
4) What are the potential impact and implication of wildlife PES on poverty and economic inequality in ASAL in general, and Maasailand in particular?
5) How has climatic variability especially in terms of drought effects affected pastoral livelihoods in the ASALs?

The rest of the paper proceeds as follows. Section 2 provides the methods, while sections 3, 4, and 5 present the results, discussion and conclusion respectively.

Methods

Data and data analyses

We compiled a database of 41 wildlife conservation initiatives in Kenyan ASALs in 2009-2011 from publically available data sources and contacts with several organizations involved in wildlife conservation, wildlife tourism, and pastoral development. This database includes community Conservancies, with some operating wildlife PES schemes (Appendix I). This preliminary inventory is by no means exhaustive as the greatest challenge encountered in this process was the paucity of the data on Conservancies in Kenya.

We obtained the data on income poverty and economic inequality for 1999 from government surveys (Central Bureau of Statistics, 2003, Central Bureau of Statistics, 2005), human population in ASALs (for 1979,1989, 1999 and 2009) from national census reports (Kenya National Bureau of Statistics, 2010, Kenya National Bureau of Statistics, 2009) and of tourism bed-nights in Maasailand (for 1999 to 2010) from the 2011 government statistical report (Kenya National Bureau of Statistics, 2011). There are however some caveats to these data. First, the official statistics for pastoralists population numbers and poverty levels need to be treated with caution because they are commonly incomplete or are of poor quality due to reasons related to remoteness, access, coverage, mobility, language differences and interviewee concerns as to tax and other implications of questions around income and assets (Randall, 2008). Secondly, the publicly available data on population and poverty is dated and is more than 10 years. Unfortunately, we were unable to obtain more recent data on the same.

In line with the official tourist zonation in Kenya, we considered Maasailand as the area in southern Kenya that covers the two Counties of Narok and Kajiado, and also include a small portion of Machako County (Kenya National Bureau of Statistics, 2011). We obtained the data on wildlife and livestock populations in ASALs from the aerial surveys conducted by the Department of Resource Surveys and Remote Sensing (DRSRS), the data on cropland areas from the Kenyan cropland distribution maps of the 1970s (KREMU, 1983) and 2000 (FAO, 2005a), and the data on rainfall and temperature trends from 1960 projected to 2025 from the Famine Early Warning Systems Network (FEWSNET: www.fews.net).

We restricted our analysis to Conservancies and wildlife PES schemes (hereafter Conservation schemes) located on community trust lands, group ranches or private land owned individually or collectively by indigenous pastoral communities. We did not include in our analysis the protected area owned and managed by local authorities on behalf of local communities but contracted to private sector entity such as the Mara Conservancy and the large private and commercial ranches and conservancies not located on community owned land such as the Lewa Conservancy.

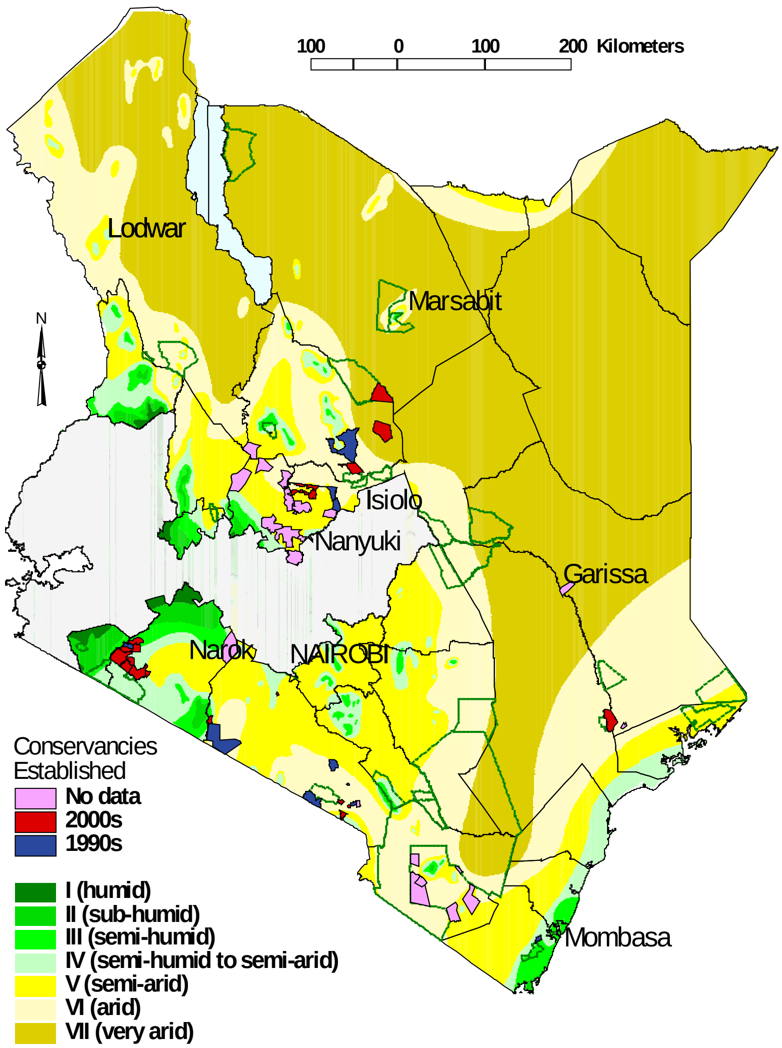
We classified these conservation schemes as follows: (1) by the nature of the production system (whether based on wildlife only or integrated livestock-wildlife system); (2) the prevailing land tenure (whether located on private land that is individually owned or on group owned land); (3) by the funding typology (whether "public" or "market" based). Public funding refers to financing by public sources, such as philanthropic donations, NGO grants, government funds and bilateral and multilateral grants. Market funding refers to self-organized deals with financing by private commercial tourist companies; and (4) lastly, by the conservation approach and directness of incentives.

We then produced land use maps in GIS showing biodiversity (wildlife biomass distribution in 2005-2010), the supply of two ecosystem services (wildlife habitat services in public protected areas and on private lands in 1990s and 2000s and, provisioning services for crop production in 1970s and 2000), and macro-economic poverty variables (the poverty rate and the annual investments required to fill the poverty gap), and the economic inequality based on the gini-coefficient (Figure 3.1).

The poverty rate (also known as "headcount ratio") is the percentage of the total population living below the 1999 Kenyan rural poverty line of US$ 0.59 per capita per day while the poverty gap (also known as the depth of poverty) represents the average expenditure shortfalls for the poor relative to the rural poverty line.

We calculated the theoretical annual financial investments required to close the poverty gap by multiplying the poverty gap by the rural poverty line and the number of poor people per unit area. The economic inequality is based on the gini-coefficient of the 1999 per capita expenditures. The gini-coefficient is a widely used measure of inequality. It varies from zero representing perfect equality, a situation where each individual or household has the same income or expenditure and one representing perfect inequality, a situation where one person or household has all the income or expenditure.

Figure 3.1 Map showing the location of conservation schemes (Conservancies and wildlife PES schemes: 1990s and 2000s) in relation to the agro-climatic zones (ACZs) in Kenyan ASALs.
Data source; The geospatial layers was obtained from ILRI (International Livestock Research Institute: ILRI) and the Conservancies from the Authors' inventory (Appendix I).



We then calculated the changes in human population density across the ASAL districts, including the two Maasailand districts of Kajiado and Narok for the 30 year period between 1979 and 2009. In Kenya, districts were the administrative unit for development planning until 2010 when these were replaced by Counties which are currently the devolved units for governance as stipulated in a new Constitution adopted in 2010. The current Counties as listed in the 2010 Constitution represents the Districts as they existed prior to 1992.

In the case of Maasailand, we first calculated the ratio of the provision of wildlife habitat services in private lands to that of public protected areas by dividing the total area allocated for wildlife conservation in private and communal land to the total area covered by wildlife protected area. Second, we used market prices to estimate the actual and potential value of tourism services based on the tourism bed-nights occupied and available respectively from 1999 to 2010. We computed the value of tourism services by multiplying the number of bed-nights in 2010 by the estimated tourist expenditure on excursion and park fees in 2007 of US$ 40.71 per person per bed-night (World Bank, 2011) and the value per hectare generated from the total area (private and public) allocated for wildlife habitat services (This includes the two protected areas of Maasai Mara National Reserve and Amboseli National Park and all the community conservancies and wildlife PES schemes in Maasailand).

We then compared the supply of tourism services to that of wildlife habitat services in private lands and estimated the potential effect of annual PES cash transfers on the poverty gap across three scenarios of US$ 10/ha (lower), US$25/ha (middle) and US$40/ha (upper). Our estimated tourism values does not distinguish between private lands and protected areas because the tourism bed-nights data used is not disaggregated by these two land management categories.

Lastly, we prepared a map of community conservancies in Kenyan ASALs and overlaid it to a climatic variability map of observed and projected changes in rainfall and temperature in Kenya during the long rainy season (March to July) for the period from 1975 to 2025 (FEWSNET, 2010). This enabled us to explore the implications of short-term changes in precipitation and temperature on conservancies in different ASAL regions of Kenya.

Results

Classification of Conservancies and Wildlife PES schemes

Three-quarters or 76% of the conservation schemes in our database are based on a "project model" and the 24% that are "non-project" based are all Wildlife PES schemes. The number of conservation schemes in Kenyan ASALs increased from one in 1992 to 41 in 2010, with much of this increase occurring between 2000 and 2010 (Figure 3.2a). Over 80% (34 schemes) operate an integrated livestock-wildlife production system and only 15% (6 schemes) maintain an exclusive wildlife and tourism production system that excludes livestock (Figure 3.2b). Many of the schemes with an integrated livestock-wildlife production system apportioned a core zone restricted to wildlife and tourism where livestock are excluded except during the dry season or drought periods when they serve as grass-banks for that time.

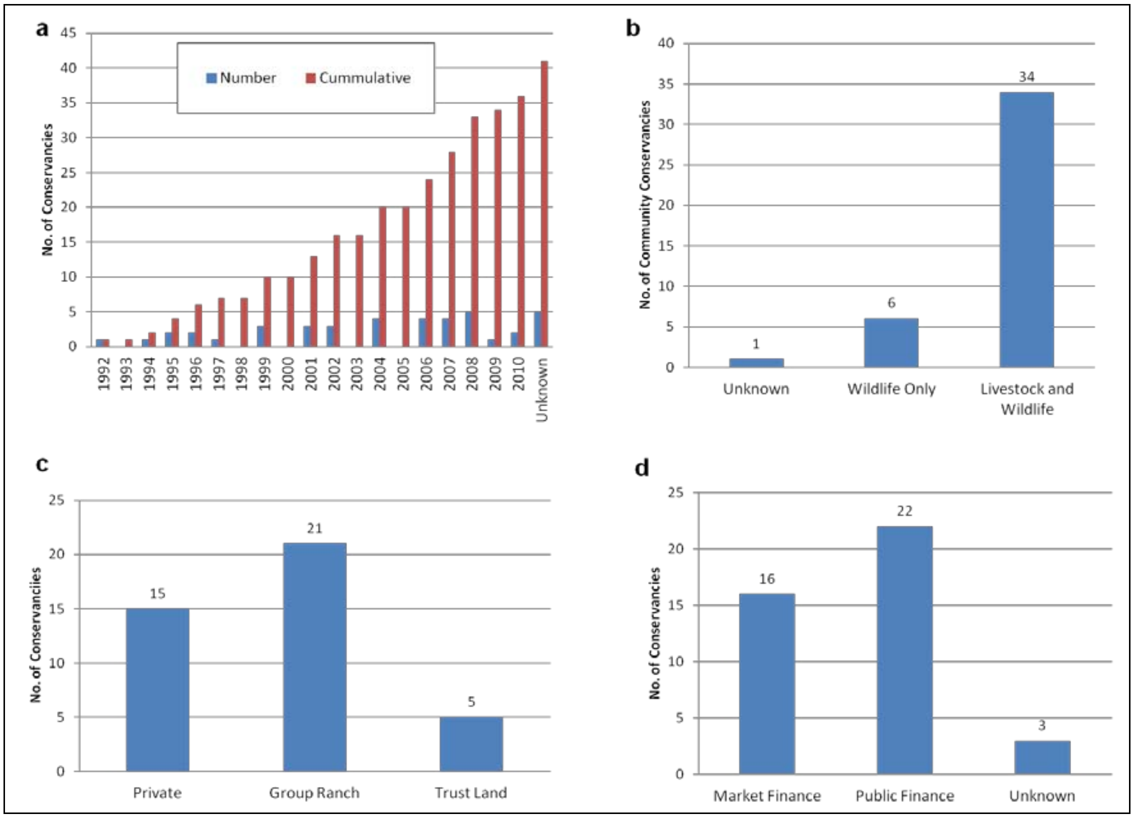
The proportion of the schemes that were funded from market sources differed between tenure types; most on group owned lands relied on public funding; significantly more on individuated private lands obtained funding from market sources (Table 3.1; $z = 3.22$, $p<0.001$).

Table 3.1 The biodiversity conservation schemes on private and communal lands in Kenyan ASAL by land tenure and funding typology.
Data source; Author's database (Appendix I)

Land tenure	Funding typology	Number of Conservancies and PES schemes	
		2000	**2010**
Communal land (group owned: Trust Lands and Group Ranches)	Market	2	7
	Public	3	14
Privatized land (individual owned)	Market	1	9
	Public	1	6

Over 65% of all the conservation schemes are located on group owned land with 51% on group ranches and 12% on trust lands. The private, individuated land accounted for 37% the conservation schemes and all are wildlife PES schemes (Figure 3.2c). More than half (54%) of the conservation schemes were funded from public sources, and another 39% from market sources. We lacked information on the nature of funding for the remaining 7% of the conservation schemes (Figure 3.2d).

Figure 3.2 A. The number of conservation schemes (Conservancies and wildlife PES schemes) in the Kenyan ASALs from 1992 to 2010; **B.** the Conservancies classified by production system; **C.** land tenure type; and **D.** funding typology.
Data source; Authors database (Appendix I)



Changes in biodiversity and supply of ecosystem services

Table 3.2 shows the Agro-Climatic Zones (ACZs) in Kenyan ASALs, the changes in biodiversity using wildlife biomass as a proxy (1970s to 2005-2010), the changes in the supply of wildlife habitat services in public (protected areas) and private lands (conservation schemes) , and the changes in provisioning ecosystem services of crop (1970-2000), and livestock (1970s to 2005-2010) production.

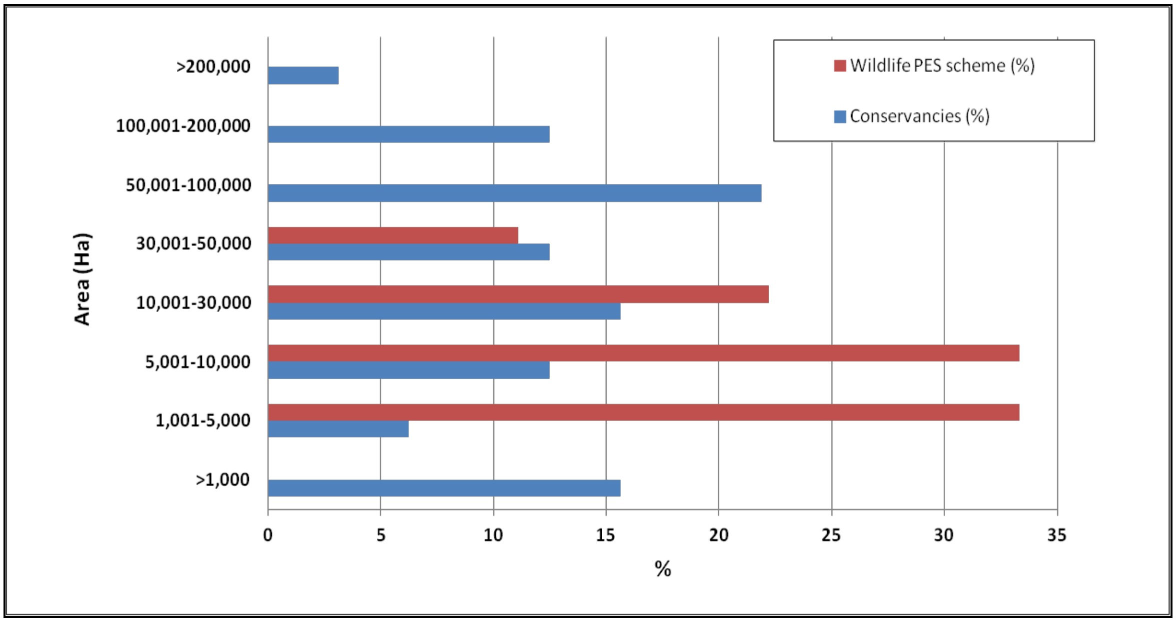
Table 3.2 Land (area in km^2 and % of total ACZ area) of Kenyan ASAL under protected areas (parks) and conservation schemes (Conservancies and wildlife PES schemes) in 2010 and under crop cultivation in the 1970s and 2000. Biomass (g.m^{-2}) of wildlife and livestock in the late 1970s and 2000s for non-park rangelands, respectively converted (Yes) or not (No) in 2000.
Data source; DRSRS; KREMU (1983); FAO (2005).

Agro Climatic Zone (ACZ)	**III-IV**		**V**		**VI-VII**		**ASAL**	
Description	Sub-humid		Semi-arid		Arid			
	Area	**%**	**Area**	**%**	**Area**	**%**	**Area**	**%**
Total ACZ area	35,645		80,667		389,274		505,586	
Protected Areas (PA's) 2010	2,395	6.7	2,496	3.1	33,451	8.6	38,341	7.6
Conservancies 2010	1,635	4.6	2,229	2.8	5,894	1.5	9,757	1.9
Cropland 1970s	13,322	37.4	23,447	29.1	5,001	1.3	41,770	8.3
Cropland 2000	16,329	45.8	29,816	37.0	9,840	2.5	55,984	11.1
Change in cropland	**3,007**	**22.6**	**6,369**	**27.2**	**4,839**	**96.8**	**14,214**	**34.0**
Rangeland converted to crops	**Yes**	**No**	**Yes**	**No**	**Yes**	**No**	**Yes**	**No**
Wildlife biomass late 1970s	1.64	11.51	1.2	1.25	0.66	0.35	1.1	0.72
Wildlife biomass late 2005-2010	0.32	1.77	0.52	0.72	0.25	0.13	0.38	0.23
Change wildlife biomass (%)	**-80.5**	**-84.6**	**-56.7**	**-42.4**	**-62.1**	**-62.9**	**-65.5**	**-68.1**
Livestock biomass late 1970s	5.24	5.65	5.55	4.47	4.21	2.12	4.98	2.43
Livestock biomass late 2005-2010	5.49	6.81	2.96	3.12	2.21	1.81	3.23	2.05
Change livestock biomass (%)	**4.8**	**20.5**	**-46.7**	**-30.2**	**-47.5**	**-14.6**	**-35.1**	**-15.6**

The supply of wildlife habitat services on private land differed among ACZs, with highest proportions found in the sub-humid zones III/IV (Table 3.2). By 2010, the total area of private land allocated for wildlife habitat services in Kenyan ASALs equaled 9,757 km^2 (975,700ha) which represents 1.9% of the total ASAL area in Kenya (Table 3.2).

Conservation schemes vary considerably in size (Figure 3.3). Conservancies are generally larger in size with 38% of such areas being >50,000ha. No single wildlife PES scheme is >50,000ha; only 11% fall between 30,000ha and 50,000ha, with the majority (66%) falling between 1,000ha and 10,000ha (Figure 3.3).

Figure 3.3 The sizes of individual Conservancies and wildlife PES schemes in Kenyan ASALs.
Data source; Authors database (Appendix I).



There are marked differences in the supply of provisioning ecosystem services in the ASAL as proxied by changes in the land area under crops and the livestock biomass. The spatial distribution of cropland varied between drier and wetter ACZ with the latter recording a higher percentage of cropland area (Table 3.2; Figure 3.4).

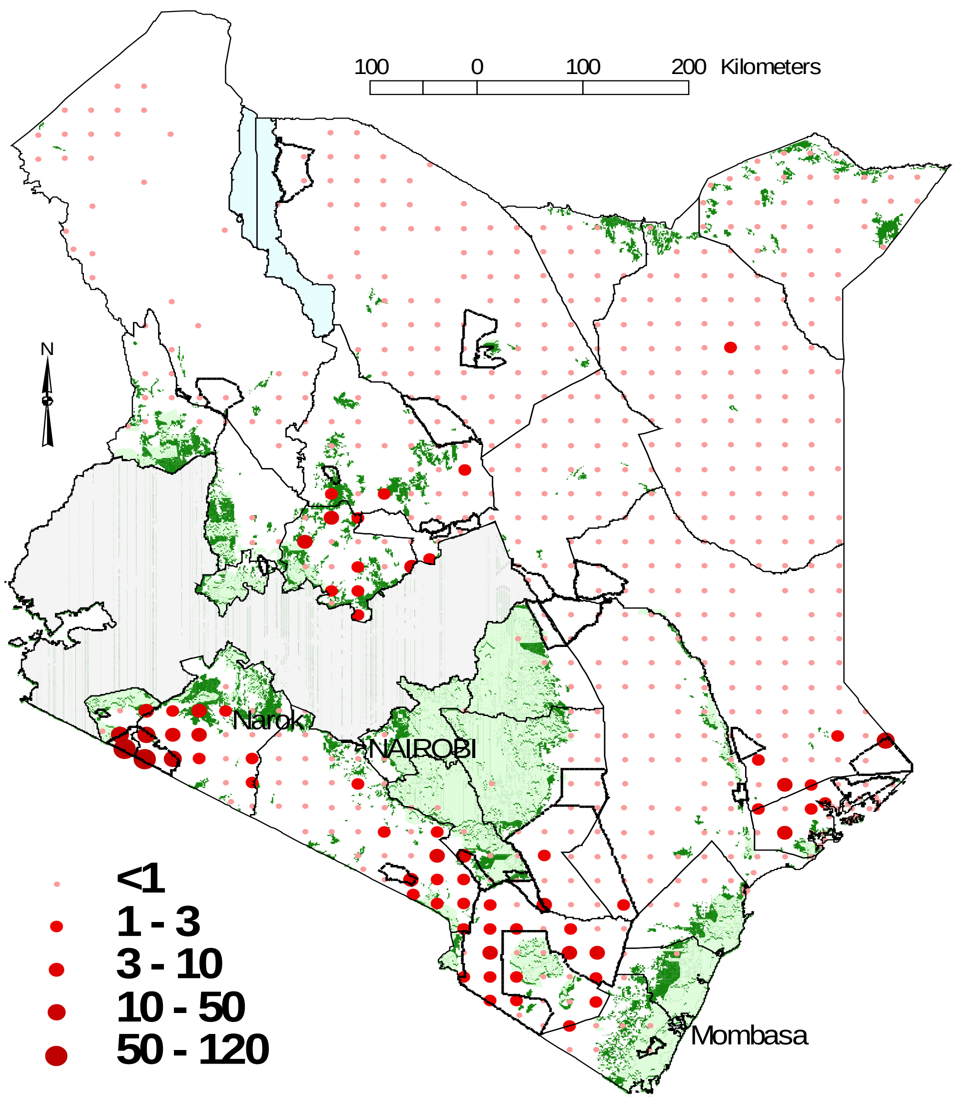
Overall, the total area under croplands increased by 34% between 1970s and 2000, with a higher proportion of the increase occurring in the drier zones (96.8% in the arid areas) and much lower rates in the wetter zones (27.2% in the semi-arid and 22.6% in the sub-humid zones: Table 3.2).

Similar variations by ACZs were also observed for livestock with the recorded biomass in the sub-humid wetter zones III/IV over twice that in the arid drier zones VI/VII. In the period between the 1970s to 2000s, livestock biomass increased both in converted (4.8%) and non-converted (20.5%) areas in zone III/IV, but declined overall in drier zones both in unconverted (30.2% in the semi-arid and 14.6% in the arid) and, more significantly, in lands converted to crop cultivation (46.7% in the semi-arid and 47.5% in the arid; Table 3.2). The recorded decline in livestock biomass in the ASALs between 1970s and 2000s was more than twice in the areas converted to cropland (35.1%) compared to non-converted areas (15.6%; Table 3.2).

Our results also show variations in wildlife biomass, by ACZs; the average wildlife biomass in 2005-2010 varied from <0.15g.m^{-2} (1.5kg/ha) in the arid zone in the North and East to >1 g.m^{-2} (10kg/ha) in semi-arid to sub-humid rangelands of Central and Southern Kenya, where pockets remain with wildlife biomass >10 g.m^{-2} (100kg/ha) and higher (Figure 3.4, Table 3.2).

Overall, ASAL wildlife biomass has declined by 65% and 68% in areas converted to croplands and unconverted respectively since the 1970s (Table 3.2). There was no significant difference in rates of wildlife decline between areas with cropland expansion and unconverted areas. In areas converted to cropland in the wettest zones, wildlife densities were already 85% lower prior to conversion compared to non-converted areas, with a further decline after conversion. Thus, the major loss of wildlife (from 11.51 to 1.77 g.m^{-2}) preceded that induced by land use change (Table 3.2).

Figure 3.4 The land use map of Kenyan Arid and Semi-Arid Lands (ASAL) showing the wildlife biomass in 2005-2010 (g.m^{-2}) and the changes in area under cropland between 1970 (light green color: ●) and 2000 (dark green color:●). The land cover accuracy in 1970s is 95%, and in 2000 is 89%.
Data source; The data used to produce the map was obtained from the Department of Resource Survey & Remote Sensing (DRSRS); Kenya Rangeland Monitoring Unit (KREMU: 1983); FAO (2005).



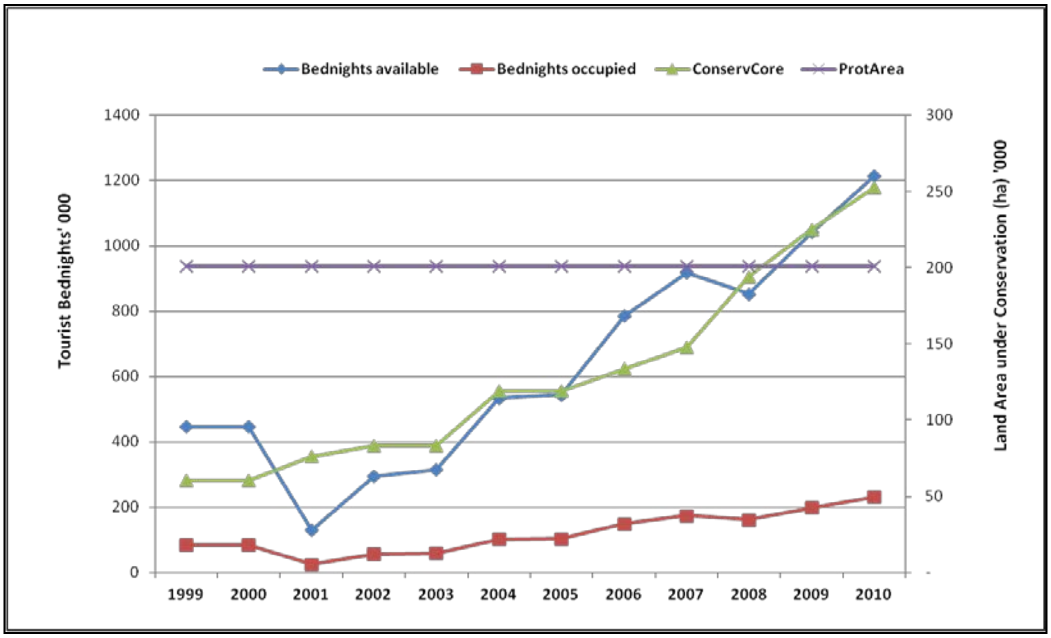
At the regional level in Maasailand, the provision of wildlife habitat services on private lands increased four-fold from an area of 60,319ha (1.5%) in 1999 to 252,613ha (6.3%) in 2010. The year 2008 marked a critical turning point when the area under private and communal land allocated for wildlife in Maasailand equaled that for public protected areas (Table 3.3.) Since then, there is now more private land allocated for wildlife habitat services in Maasailand than in public protected areas, with a current difference of some 26% (Table 3.3; Figure 3.5).

Table 3.3 The proportion of private land area under conservation schemes (Conservancies and wildlife PES schemes) (%), and the ratio of private land area under conservation schemes to protected areas in Maasailand.
Data source; Author's database (Appendix I); KNBS (2011).
The protected areas include the Maasai Mara National Reserve (MMNR - 150,000ha) in Narok County; Nairobi National Park (NNP- 11,700ha at the northern edge of Kajiado County); and Amboseli National Park (ANP -39,200ha) in Kajiado County

Year	**Private land in conservation schemes (%)**	**Ratio of conservation schemes to protected areas**
1999	1.5	0.3
2000	1.5	0.3
2001	1.9	0.4
2002	2.1	0.4
2003	2.1	0.4
2004	3.0	0.6
2005	3.0	0.6
2006	3.4	0.7
2007	3.7	0.7
2008	4.9	1.0
2009	5.7	1.1
2010	6.3	1.3

Correspondingly, the actual and potential provision of tourism ecosystem services in Maasailand using the proxies of the tourism bed-nights occupied and available respectively also increased over the same period, except for 2000-2001. The recorded increase in the supply of tourism ecosystem services was highly correlated with the increase in the supply of wildlife habitat services on private lands in the period 1999 to 2010 ($r^2 = 0.92$; Figure 3.5). However, the supply outstripped the demand for tourism ecosystem services, with the supply-demand gap progressively increasing from 238,000 tourism bed-nights in 2002 to 938,000 tourism bed-nights in 2010 (Figure 3.5). This gap shows that at a regional level, investments in increasing tourism facilities in Maasailand is not matched by a corresponding increase in tourism visitations.

Figure 3.5 The supply (bed-nights available) and demand (bed-nights occupied) for tourism, and the state (protected area) and private (ConservCore) land under conservation in Maasailand for the period 1999 to 2010.
Data source; Authors database (Appendix I); KNBS (2011)



Demographic changes in ASALs

The official population statistics show that overall, the total human population in the ASAL increased three-fold; from 3.62 million people in 1979 to 9.25 million people in 2009. With the exception of the 1979-89 decade, the rate of population increase was higher in the arid than in the semi-arid districts (Table 3.4). The current average population density in ASALs is 20 people per km^2, (0.2 people/ha) but this varied in the arid and semi-arid districts. The density in arid districts increased four-fold from three persons per km^2 in 1979 to 14 persons per km^2, in 2009, while in the semi-arid districts, the density increased only two-fold from 15 persons per km^2 to 30 persons per km^2.

Regionally, the recorded rate of human population change in Maasailand, which falls in semi-arid districts, was highest at 9.3% per annum between 1979 and 1989, but reduced to 3.6% and again increased to 6.4% between 1989-1999 and 1999-2009 respectively (Table 3.4). The population density in Maasailand also increased fourfold from 10 persons per km^2 in 1979 to 40 persons per km^2, in 2009, exceeding that of all the semi-arid districts of 30 persons per km^2 in 2009 (Table 3.4).

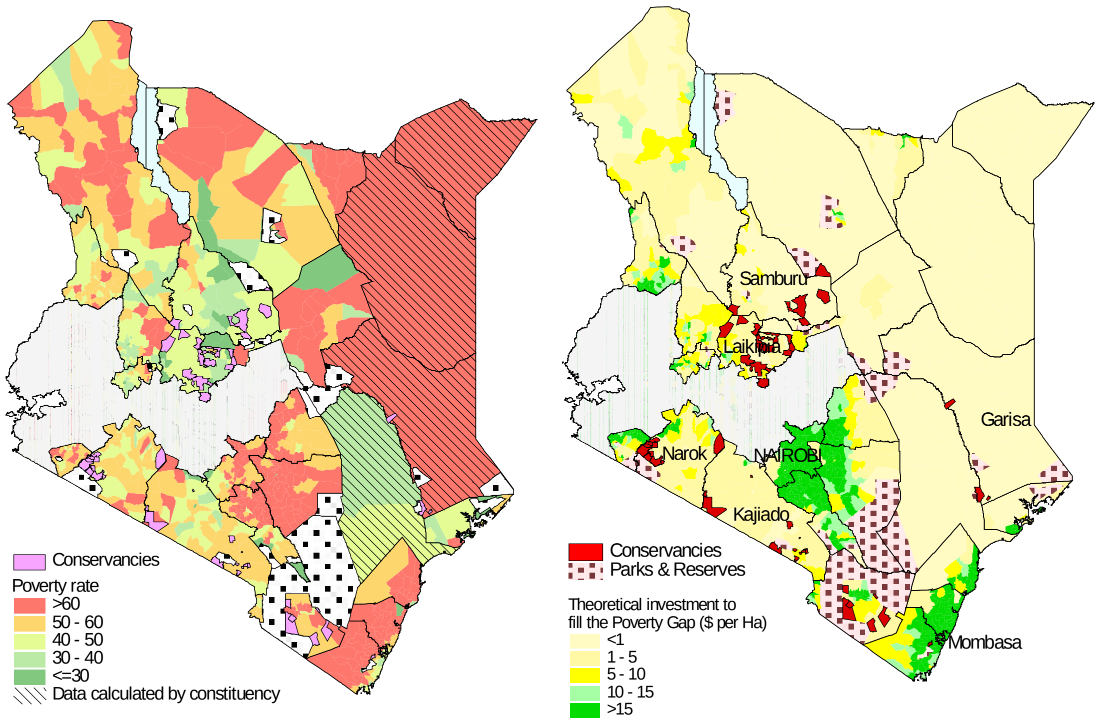
Table 3.4 The human population growth rate in arid districts, semi-arid districts, and in Maasailand for the periods 1979-1989, 1989-1999 and 1999-2009, with population density (number of people/ha) for 1979, 1989, 1999 and 2009 in brackets and italics.
Data source; KNBS (2009; 2010; 2011)
The list of the Arid and semi-arid Districts and the map of the respective Counties in which they fall is provided in Appendix I-B and I-C respectively. Maasailand consists of Kajiado County and Narok County (inc. Trans-Mara District).

	Average annual population increase (%)			
ASAL regions	**1979**	**1979-1989**	**1989-1999**	**1999-2009**
Arid Districts	*(0.03)*	3.0 *(0.04)*	6.2 *(0.07)*	10.3 *(0.14)*
Semi-Arid Districts	*(0.15)*	5.1 *(0.23)*	1.2 *(0.25)*	3.1 *(0.33)*
Maasailand[3]	*(0.09)*	9.3 *(0.17)*	3.6 *(0.24)*	6.4 *(0.40)*

Income poverty assessment: poverty rate, poverty gap and the theoretical investments required to fill the poverty gap

Our results indicate a correlation between the ASAL areas where conservation schemes are located to areas recording moderate (between 40-60%) to high (>60%) poverty rates (Figure 3.6a). Theoretically, the amount of money per unit area needed to fill the poverty gap across the ASAL regions is generally low, being < US$ 15/ha in many of the areas (Figure 3.6b). The only exceptions are parts of the semi-arid lands to the East of Nairobi metropolis and the semi-humid areas along the coastal strip, around Mombasa city, areas located to the West of Narok County, and small patches to the West of Samburu County located on the Kenya-Uganda border, where > US$ 15/ha investments is required to fill the poverty gap (Figure 3.6b).

Figure 3.6 The map of Kenyan ASAL showing **A.** Poverty rate and **B.** The theoretical investment (US$.$ha^{-1}.yr^{-1}$) required to fill the poverty gap.
Data source; Conservancies data taken from the authors' database (Appendix I); Data on poverty from the Kenya National Bureau of Statistics (KNBS).



Regionally in Maasailand, the actual and potential aggregate value of tourism in 2010 is estimated at US$9.4 million and US$ 49.4 million respectively. This means that the current oversupply of tourism ecosystem services is worth some US$ 40.03 million in 2010 prices (Table 3.5). Nominally, the current tourism value is only sufficient to meet the conservation costs under the lower PES scenario of US$ 10/ha, but annual subsidies of US$ 4/ha and US$ 19/ha would be required to realise the conservation costs projected for the middle (US$25) and upper (US$ 40) PES scenarios respectively (Table 3.5). If the value of the current tourism potential were to be realised, it would generate sufficient revenues to meet the costs of conservation projected across all the three PES scenarios (Table 3.5).

Overall, in Maasailand, the amount of money generated from the actual tourism value per hectare of conserved land exceeds the cash transfers required to close the poverty gap by US$ 11/ha while that for potential tourism value would exceed the cash transfers required to close the poverty gap by US$ 102/ha (Table 3.5).

Table 3.5 The supply and demand of tourism in private and state (protected area) land in Maasailand, in relation to the costs of conservation and the potential impact of PES on poverty across three scenarios.
Data source; Author's database (Appendix I); KNBS (2011).

		PES Scenario		
	Unit	**Lower**	**Middle**	**Upper**
A: CONSERVATION				
A1: PES Cash Transfer	($/ha/yr)	10	25	40
A2: TotConArea (TCA) (2010)	(ha)	441,813	441,813	441,813
A3: Total Cost of Conservation (TCC)	($)	4,418,134	11,045,336	17,672,538
B: TOURISM DEMAND (2010)				
B1: Total Tourism Demand (TTD)	($)	9,390,169	9,390,169	9,390,169
B2: ConCostGapDD (TTD-TCC)	($)	4,972,034	(1,655,167)	(8,282,369)
B3: TTD per ha (TTD/TCA)	($/ha)	21	21	21
B4: ConCostGapDD (TTD-TCC)/ha	($/ha)	11	(4)	(19)
C: SUPPLY SIDE				
C1: Total Tourism Supply (TTS)	($)	49,421,940	49,421,940	49,421,940
C2: ConCostGapSS (TTS-TCC)	($)	45,003,806	38,376,604	31,749,402
C3: TTS per ha (TTS/TCA)	($/ha)	112	112	112
C4: ConCostGapSS (TTS-TCC)/ha	($/ha)	102	87	72
D: POVERTY IMPACT				
D1: PovGap	($/ha)	10	10	10
D2: TTD -PovGap	($/ha)	11	11	11
D3: TTS-PovGap	($/ha)	102	102	102

Notes:
A2: The sum of protected area land (Maasai Mara National Reserve and Amboseli National Park) and the private lands allocated to Conservancies and wildlife PES schemes in Maasailand
A3: Calculated as A2 multiplied by A1

B1: Calculated as tourism bed-nights occupied multiplied by the excursion and park fees of US$ 40.71/person/bed-night
B2: Calculated as B1 minus A3
B3: Calculated as B1 divide by A2
B4: Calculated as B2 per hectare

C1: Calculated as tourism bed-nights available multiplied by the excursion and park fees of US$ 40.71/person/bed-night
C2: Calculated as C1 minus A3
C3: Calculated as C1 divide by A2
C4: Calculated as C2 per hectare

D1: The theoretical investments required to fill poverty gap in Maasailand
D2: Calculated as B1 minus D1
D3: Calculated as C1 minus D1

Economic inequality

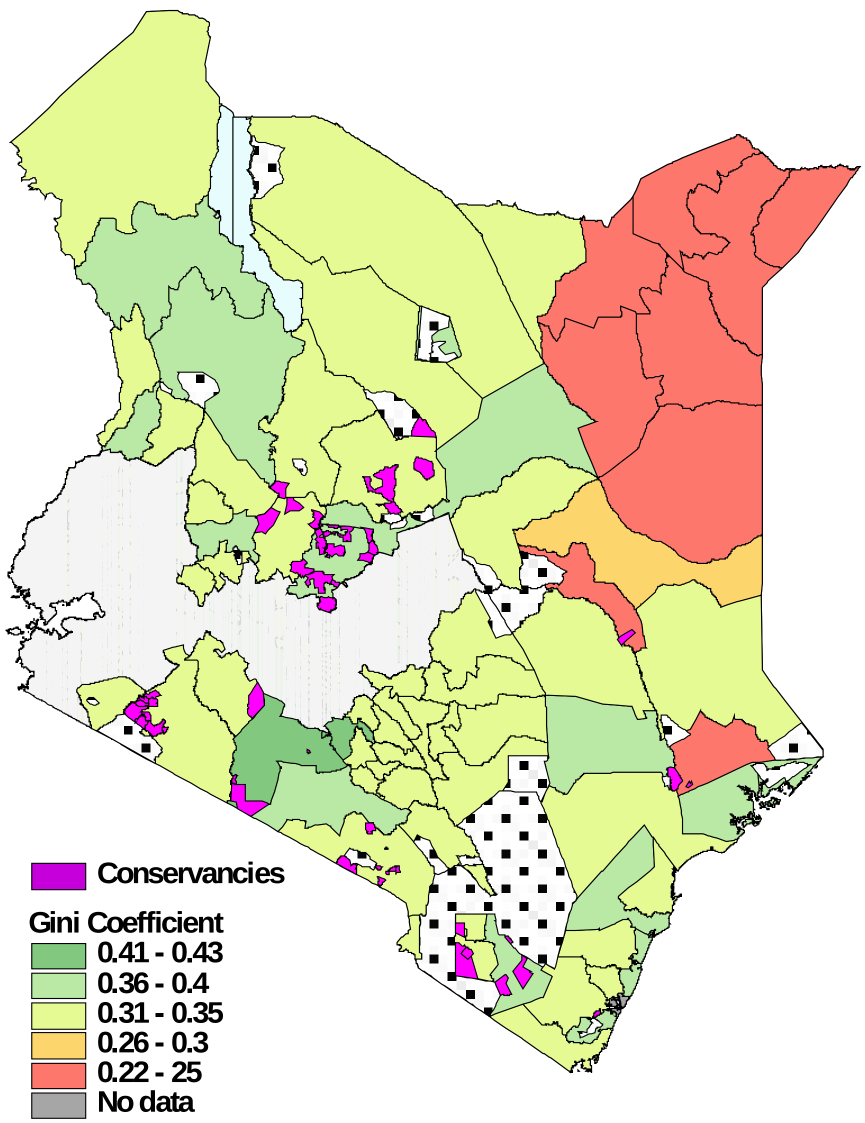
Economic inequality can be expressed in terms of the levels of differences based on income and expenditure patterns, but also in terms of access to assets such as land and livestock. Here we report on the economic inequality assessment based on the expenditure patterns overall, large areas of the ASALs record gini-coefficient values of 0.31-0.35. The only exceptions are found in the Central and Northern Kenya regions which have gini-coefficient values in the range of 0.36-0.40 (Figure 3.7).

Within Maasailand, some regions have the highest levels of inequality. These include sections of Kajiado County in which three conservation schemes - Shompole and Ol'Kiramatian Conservancies and the Wildlife Lease Program in Athi-Kaputie Plains - are operational. These areas record the highest gini-coefficient values in the range of 0.41-0.43 (Figure 3.7).

Figure 3.7 Map of Kenyan ASAL showing the average inequality of per capita expenditure for 1999 as measured by the gini-coefficient for all the 210 constituencies in relation to location of protected areas (the dotted uncolored zones) and location of the Conservancies.

The grey zone that forms a belt in central and western parts of Kenya are the humid areas and not part of ASALs.

Data source; Conservancies data from the authors' database (Appendix I); Data on poverty from the KNBS



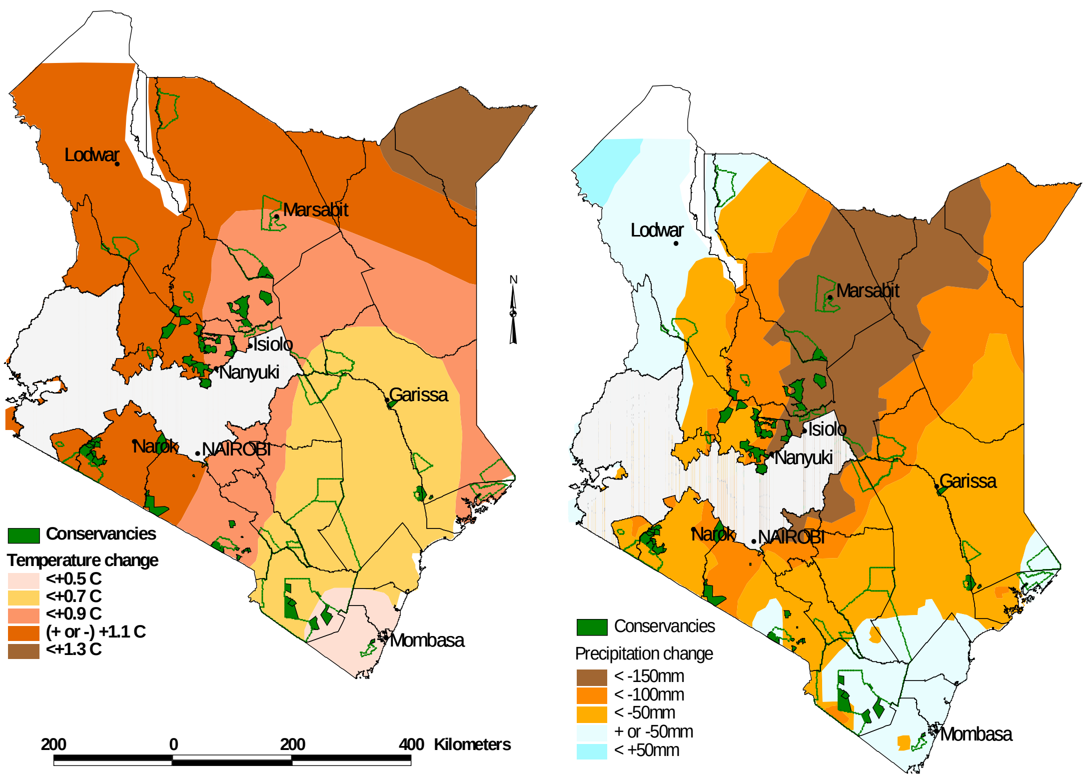
The climatic variability and pastoral livelihoods in the ASALs

Limited information on climate change is available at the country and local levels but it is acknowledged that the effects of climate change in Kenya will alter the weather patterns due to the changes in precipitation and temperature in ways that are likely to significantly affect the ASAL regions and the pastoral livestock production therein (Government of Kenya, 2010).

While the long term impacts of climate change remain uncertain, recent analyses of observed and predicted patterns of climate change based on short-term rainfall and temperature variables provide some insights already on what could be the conditions. Increases in temperature have been recorded for the period 1960 to 2009 (Funk et al., 2010) suggesting a warming trend overall (Government of Kenya, 2010), that is expected to continue to 2025 (Figure 3.8a).

There is considerable uncertainty regarding the climate change effects on precipitation in Kenya. Although there is a recorded decline in the long season rainfall from March to June (MAMJ) for the period from 1960 to 2009, a trend which is expected to continue with large parts of Kenya expected to have experienced a decline of more than 100mm in the long season rainfall between 1975 to 2005 (Figure 3.8 b) (Funk et al., 2010).
In the short-term to 2025, there is some variability in the predicted changes in temperature and precipitation across Kenya as shown in Figure 3.8.

Figure 3.8 The map of Kenyan ASAL showing the location of Conservancies in relation to observed and projected change in **A.** temperature and **B.** rainfall, for the period 1975 to 2025.
Data source; Conservancies data taken from the authors' database (Appendix I); Data on temperature and rainfall (FEWSNET 2010).



Discussion

Our analysis shows that the model of conservation schemes adopted is influenced by the prevailing land tenure regime including whether funding is provided through market or public sources. Wildlife PES schemes are only operational in privatised adjudicated and sub-divided lands. Seven of the 10 wildlife PES schemes in our database are market funded by commercial tourist operators while the remaining three are all publicly funded.

Implications for biodiversity and ecosystem services

Our analysis shows that the provisioning services of crop and livestock production (in wetter zones) have been enhanced in Kenyan ASALs since early 1970s, but at the expense of biodiversity as recorded by the declines in wildlife populations especially in the wetter ASAL areas. The declines in biodiversity are further compounded by the high human population growth rates. On average, the human population density in pastoral areas are already above the 10 people per km^2 threshold which leads to steep declines in wildlife populations (Reid et al., 2003).

Although there is no comprehensive assessment of the impact of conservation schemes on biodiversity evidence is emerging that the provision of wildlife habitat services in private lands does affect biodiversity positively (Langholz and Krug, 2004). Conservancies protect endangered species and critical natural habitats, including wildlife dispersal areas and migration corridors located adjacent to protected areas, that otherwise risk being lost to land uses that are incompatible with wildlife. Preliminary assessments reports indicate that not only are wildlife populations higher in Conservancies than in protected areas, but these populations are increasing in the former while declining in the latter (Western et al., 2006). Thus, the close to one million hectares of the ASALs under community conservation schemes constitutes a critical part of the growing private land protected for biodiversity in Kenya (Carter et al., 2008).

Livestock production was limited by both the cropland expansion and conservancy land allocated for wildlife and tourism. Our analysis showed an overall decline in livestock biomass in the ASALs, but with differences recorded for the wetter sub-humid zones, which experienced an increase as opposed to the drier semi-arid and arid areas which recorded declines. Furthermore, livestock production performed poorly in rangelands converted to crops compared to non-converted areas a trend also noted in a previous study (Norton-Griffiths and Said, 2010).

Although we do not yet know the large scale effect of conservation schemes on livestock production, our analysis showed a significant variation in the degree of integration of livestock and wildlife. While in some conservancies, livestock grazing is totally excluded except during dry seasons or drought periods, in others, pastoralists continue to use part of the conservancy to graze livestock away from the core area set aside for wildlife and tourism facilities. And in others, reduced livestock densities are used to manage the land for wildlife.

The provisioning ecosystem services for food and fiber supply in ASALs have also been enhanced. Our analysis revealed significant cropland expansion in ASALs overall, with higher proportional increases in drier compared to wetter zones potentially because of limited land availability in the wetter zones. This makes apparent several synergies and trade-offs in the supply of ecosystem services considered (Power, 2010).

First, within the provisioning services, there exists a trade-off between livestock and crop production. The expansion of croplands leads to declines in livestock numbers as crop areas increases not only because of total area but also the impacts of fragmentation and excision of key resources making livestock less well able to make use of the remaining areas. However, the observed decline in livestock in the drier zones, which was more pronounced in the converted areas than in non-converted areas, also suggests that, in addition to cropland expansion, other drivers of change could be responsible for the declines in livestock in the non-converted drier zones. One potential driver is drought which has increased and intensified in regularity causing high livestock mortality in many parts of the Kenyan ASALs (Nkedianye et al., 2011).

Second, there exists a trade-off between crop production and the provision of wildlife habitat services (Serneels and Lambin, 2001). However, the lack of direct link between the final quasi-collapse of wildlife (from 1.64 to 0.32 $g.m^{-2}$) and conversion to crops suggests that prior processes of land fragmentation and loss of access to key resource such excision of wetlands (Scoones, 1991), and dry season pasture could probably be more important drivers of wildlife decline in ASALs than cropland expansion *per se*.

Third, there are spatial synergies among the provisioning service of livestock production, the cultural service of pastoralism as a tradition and wildlife tourism and recreation, and the provision of wildlife habitat service within an integrated livestock wildlife production system (Ericksen et al., 2012). .

Implications for pastoral poverty livelihoods

Analysis of the 41 conservation schemes included in our database showed that a total of 98,963 people are directly involved either as landowners enrolled in the wildlife PES schemes in areas of individually owned land, or as members of Conservancies in areas of group owned lands. We however speculate that this number could be higher if one considers the numbers of people that are also affected or impacted indirectly by these conservation schemes. To illustrate, a total of 589,400 ha of land are currently under these conservation schemes in the arid zones (ACZ V-VII) and with a population density of 0.14 people/ha, so roughly 82,500 people are likely impacted by these schemes. Another 386,400 ha are under conservation schemes in the semi-arid zones (ACZ III-V) with a population density is 0.33 people/ha, so the roughly 127,500 people are impacted by these schemes. In total, we speculate that an estimated 210,000 pastoral people are directly and indirectly impacted by the existing Conservancies and wildlife PES schemes. This figure translates to 2.3% of the total Kenyan ASAL population.

The estimated figure for the number of people that are impacted by conservation schemes does not however reveal the direction of the impact which can be negative and thus considered as cost or positive and thus considered as a benefit. It also does not quantify the magnitude or level of the costs and benefits which landowners and other community members incur, and whose implications are highlighted in the next paragraph.

The costs associated with wildlife conservation will include the direct costs such as staff, infrastructure and management costs, the wildlife costs to other economic activities such as damage to crops and predation of livestock, the competition for water and disease transmission, and the wildlife opportunity costs which is the cost of alternative land uses and earnings foregone. On the other hand, the wildlife benefits to communities include the use values such as the direct values involving the consumption of wildlife products, the indirect values of the ecological and environmental services provided by wildlife, and the option value which is the premium associated with wildlife conservation for future uses. The benefits also include non-use values expressed as existence value which refers to intrinsic value attached to wildlife regardless of its use (Emerton, 1999).

These range of costs and benefits implies the need to move beyond the benefit based models of conservation to the economics of conservation for local communities and landowners (Emerton, 1999, Norton-Griffiths and Said, 2010). A comprehensive cost-benefit analysis is therefore recommended as a research priority in the future.

From a livestock based asset poverty dimension, there is a need to assess the different potential outcomes of conservancies and PES scheme by distinguishing *transitory* from *chronic* poverty. The *transitory poverty* refers to a situation whereby pastoral a households falls temporarily into poverty but retain the ability to move out again either on their own or aided by traditional safety nets. *Chronic poverty* on the other hand refers to a situation whereby a pastoral household that is locked into poverty in the long-term is unable to escape without external assistance through policy interventions (Barrett, 2005).

Livestock assets are critical in determining the nature of poverty among pastoral households. Costs associated with loss of livestock assets such as wildlife predation and disease transmission can serve to reduce pastoral household's per capita livestock holdings leading to transitory poverty in the short-term and even to chronic poverty in the long term.

On the benefit side, the ecosystem services analyzed here provide a range of local, domestic and global benefits to both local and remote populations with different implications for poverty reduction among pastoral landowners as ecosystem service providers. The *habitat services* are considered as intermediate services which are non-marketed but does provide global benefits as public goods (TEEB, 2010). Consequently, there is no direct demand for habitat services, so the payments to pastoral landowners may have to be sourced from national governments (e.g. through protected area and wildlife management agencies), and from north-south financial transfers through bilateral and multilateral programmes (e.g. the Global Environment Facility grants) and Conservation NGOs. In ASAL areas with wildlife tourism potential (e.g. adjacent to protected areas), the PES payments can be derived through user-financed PES schemes funded by tourism enterprises.

The benefits of both livestock and crop production are captured at the local level, directly contributing to poverty alleviation locally through food supply and cash income (Table 3.6). Unlike the final ecosystem services of livestock and crops, habitat services is an intermediate service that produces the final service of wildlife (Fisher et al., 2008). This generates food, ecotourism and income benefits, which are captured at all scales. The demand for final services such as tourism and recreation which are dependent on wildlife habitat services as an intermediate service is mainly from the private sector in the tourism industry, trophy hunters (where consumptive utilization of wildlife is allowed) among others, and from the public sector that is interested in the provision of biodiversity as a public good.

Table 3.6 The intermediate and final ecosystem services, the types of benefits generated the scale at which the benefits are captured, and the poverty implications among pastoral ecosystem service providers in Kenyan ASALs.
Source: Author's summary based on qualitative assessment

Ecosystem Services (ES)		**Type of benefits**	**Scale of benefit capture**			**Poverty implications for ES providers**
Intermediate Ecosystem Service	**Final Ecosystem Service**		**local**	**national**	**global**	
Provisioning Ecosystem Service						
Livestock production	Livestock	Food, assets, income	+			Marketed ES: Financial income,

						livestock per capita
Crop production	Crops	Food, assets, income	+			Marketed ES: Financial income and diversification
Habitat services						
Maintenance of life cycle of migratory species	Wildlife species	Food, eco-tourism, income	+	++	+++	Non-marketed ES: potential for public PES
Maintenance of genetic diversity	Wildlife species	Food, eco-tourism, income	+	++	+++	Non-marketed ES: potential for public PES
Cultural and amenity services						
Tourism and recreation	Wildlife species and landscapes	Ecotourism, income	+	++	+++	Marketed ES: potential for private sector PES
Cultural heritage	Socio-cultural values	Pastoral identity, cultural tourism, income	+	++	+++	Marketed ES: potential for private sector PES; "safety-net" role of social solidarity

Key: hypothesized level of importance by scale - **+++:** high; **++:** medium; **+:** low

PES is as a policy instrument that can allow pastoral landowners to capture the benefits for the provision of wildlife habitat services on their private lands. It can provide cash transfers directly to households conditional on specified ecosystem management goals to improve the environment, in a similar fashion as are conditional cash transfers to social goals that improve human welfare, both of which are critical for poverty reduction (Rodriguez et al., 2011b). Here we consider the potential of such pro-poor benefits of wildlife PES in pastoral areas using the Maasailand for illustration, and review how wildlife PES schemes across Kenyan ASAL might be funded.

In regard to analyzing the potential for PES to reduce income poverty in pastoral areas from a macro-economic perspective, the "poverty gap" approach, because it combines the headcount number with the distance from the poverty line, is a suitable indicator for which to compare the magnitude of PES transfers per unit area of land to the amount of financial investments per unit area of land required to lift the poor above the poverty line. It is thus directly relevant to policy at the macro-economic level since it measures the minimum cost for eliminating poverty through transfers. Our analysis shows that despite the high poverty rates, low population densities mean relatively low annual per hectare cash transfer of around US$ 10 to US$ 15 would theoretically suffice to close the poverty gap across much of the ASALs. Consideration of the opportunity costs of PES may increase these estimates however.

All the wildlife PES schemes in our database are located in Maasailand, where results of simulated predictive models on the potential impact of PES on poverty have shown that PES can concurrently provide wildlife habitat services while reducing poverty among the pastoral Maasai landowners (Bulte et al., 2008a). The existing wildlife PES schemes provide annual payments to pastoral landowners at rates that range from US$12/ha in publicly funded schemes (AWF, 2009, Rodriguez et al., 2011a) to US$ 43/ha in user-funded private schemes (Aboud et al., 2012a). Given that the differential returns to land from livestock, agriculture and wildlife are strongly influenced by annual rainfall availability, higher PES rates would be required in the wetter semi-arid and humid zones to induce pastoral landowners to choose the provision of wildlife habitat services over livestock and crop production (Norton-Griffiths and Said, 2010). We thus considered three PES scenarios based of lower (US$ 10/ha), middle (US$ 25/ha) and upper ((US$ 40/ha) along the ACZ gradient.

Our analysis shows that in Maasailand, the actual and potential market value of tourism on a per hectare basis in 2010 is of a magnitude considerably higher than the US$ 10/ha amount required to close the poverty gap across all the three PES scenarios. The actual market value of tourism can generate an annual surplus cash income of US$ 11/ha while that for the potential market, a surplus of US$ 102/ha. These benefits of tourism however accrues both to private landholders involved in conservancies and to the state through national parks, but the data on tourism bed-nights does not distinguish whether these are in state protected areas or in the private and communal lands.

The huge and growing demand-supply gap in tourism bed-nights is of concern for the sustainability of these conservation enterprises and also calls into the question the overhyped role of tourism in conservation. Two implications are pertinent here. One, stakeholders should find ways and means to bridge this demand-supply gap in order to realize the potential market value of tourism. Second, it suggest that the expansion of tourism facilities in Maasailand must be carefully planned and that increased use of tourism as a vehicle for promoting conservation as simplistically advocated by some conservation NGOs may not be a panacea across all areas of Maasailand. Additional investments in tourism to increase the supply of tourist bed-nights in Maasailand can only aggravate the current over-supply, reducing actual tourism values per hectare and undermining tourism's potential for poverty reduction. Beyond these, it is also necessary to ensure that the tourism revenues are directly channeled to pastoral landholders through mechanisms such as PES because this is not currently the case in many of the conservation schemes that are mostly beset with elite capture.

Our analysis has provided insights on the potential of PES to contribute towards both wildlife conservation and poverty reduction in the Kenyan ASALs. These two key goals are outlined in the national development blueprint, the Kenya *Vision 2030* which places a heavy focus on poverty reduction in the ASALs, support for the tourism industry, and the conservation of wildlife migratory corridors and dispersal areas through support towards the establishment of conservancies (Government of Kenya, 2008a). A practical question from a policy perspective is how to fund wildlife PES in ASAL to realise these dual goals of wildlife conservation and poverty reduction. As is shown from our analysis, the private sector in the tourism industry can promote the establishment of wildlife PES schemes but their reach is limited. Currently private sector investments in wildlife PES are restricted mostly to areas with significant wildlife biomass hence potential for tourism and also mostly in privatised, individually owned pastoral lands.

The areas that are wildlife rich with tourism activities only cover 5% (20,000 Km^2) of the ASALs (Norton-Griffiths and Said, 2010). The implementation of wildlife PES to supply wildlife habitat services in private lands in the remaining 95% of the ASAL currently without viable commercial tourism activities and much of it still under group ownership, will therefore fall on the public sector (GEF, 2008, Rodriguez et al., 2011a). The funding for wildlife PES schemes in these areas will need to be sourced from the public agencies; multilateral and bilateral institutions, government departments and conservation NGOs acting as PES intermediaries (Engel et al., 2008). A publicly funded wildlife PES programme can be a means to link poverty reduction and ecosystem management goals (TEEB, 2009).

These ASAL based ecosystem services will also be affected by climatic variability including changes in temperatures and precipitation. In the short term, temperature increases are expected in Kenya leading to a warming effect, accompanied by alterations in precipitation patterns, with likely implications for increased drought and flooding events. Adaptation to climatic variability in the short term and long term climate change is therefore critical for pastoral communities in the ASALs (Galvin, 2009), and as a measure for addressing the vulnerability to climate change that can lead to increased pastoral poverty (Thornton et al., 2006).

Some limitations apply to our analysis of the potential of PES on poverty in pastoral areas. First is the challenges presented by data quality and reliability. The human population and poverty data covering the pastoral areas are contestable because of the various limitations to population census and data collection in the ASAL areas. Furthermore, the publicly available data for these variables is dated and may not preset the current state of affairs. Second, , our spatial approach to data analysis masks the inherent marginalisation and other economic risks to poor pastoral households that may result from the tourism industry (Southgate, 2006, Akama, 1999) In addition, urban labour migration and other mobility associated with nomadic and semi-nomadic pastoral communities could undermine associations between spatial and social data. Third, we do not account for the income stratification within pastoral communities and the distributional inequities that may arise from any new injection of income from wildlife tourism (Thompson and Homewood, 2002, Rutten, 2004).

Conclusion

This paper uses the ecosystem services framework to explore the potential for wildlife PES to alleviate income poverty in the pastoral ASALs in Kenya, in relation to changes in wildlife biomass, selected ecosystem services, land use, demography and climatic variability. There was a mixed outcome for supply of provisioning services with recorded increase in crop production but declines in livestock production, except in the wetter semi-arid and sub-humid areas. A massive loss of biodiversity was recorded between 1970s and 2000s as wildlife biomass declined by 68% and 66% in ASAL areas with and without crop production respectively, suggesting negative impacts of land use change. The supply of wildlife habitat services on private pastoral lands increased substantially, and currently close to a million hectares of land, comprising 2% of the ASAL area is allocated to provision of wildlife habitat services. In Maasailand the supply of tourism services increased four-fold between 1999 and 2010, concurrent with the supply of wildlife habitat services in private lands, which now accounts for 6.3% of the land area under conservation and exceeds the protected area land by some 26%. The demand for tourism only increased marginally by contrast leading to over-supply of tourism services which can potentially undermine conservation schemes in the long run, and also calls into the question the overhyped role of tourism in conservation.

An estimated 210,000 pastoral people, or 2.3% of the total Kenyan ASAL population are affected directly and indirectly by the Conservancies and wildlife PES schemes assessed generating both benefits and costs which need to be assessed comprehensively. The correlation between areas of high poverty rates and high wildlife biomass suggests a potential for wildlife PES to impact on poverty. Without accounting for the opportunity costs, an estimated annual PES transfer of US$ 10-15 per hectare would be sufficient to close the poverty gap in much of the ASALs. While much of this funding can be provided by the private sector in the tourism industry, their operations are limited to only 5% of the ASAL area. It implies that currently, wildlife PES schemes in a large portion of pastoral lands can only be supported by the public sector especially through government funding. A government wildlife PES scheme can be a means to link poverty reduction and ecosystem management as outlined in Kenya *Vision 2030*, and is worth consideration.

The effects of climate change are already being experienced in Kenyan ASALs with recorded increases in temperature and declines in long season rainfall in the period from 1960 to 2009. These trends are expected to continue in the future short term to 2025, altering climate patterns that will affect pastoral livelihoods through increased drought and flooding events. There is considerable uncertainty in the long term concerning the effects of climate change. While temperatures are expected to increase, the variability in rainfall patterns remains unclear.

BRIDGE BETWEEN CHAPTERS 3 AND 4

Chapter 3 explored the potential for wildlife PES to alleviate poverty in the ASALs in Kenya in relation to changes in biodiversity (using wildlife biomass as a proxy), ecosystem services, land use, demography and climate variability. In addition, the chapter assessed the supply of tourism services wildlife habitat services on private lands between 1999 and 2010. It then estimated the number of people affected or impacted directly and indirectly by the existing wildlife PES in Kenyan ASALs, the potential effect of PES on poverty at the macro-level and the possible mechanism for funding PES schemes. Chapter 3 found that there was an overall decline in wildlife biomass, a corresponding increase in the ASAL area under crop production, and an overall decrease in livestock production in ASAL areas (except for the wetter sub-humid zones) between 1970s and 2000. In addition, the supply of wildlife habitat services in private lands increased between 1992 and 2010. It showed that in Maasailand, the demand for tourism services only increased marginally but the supply of tourism services increased four-fold, concurrent with the supply of wildlife habitat services in private lands.

Chapter 3 concluded that close to 2.3% of the total Kenyan ASAL population are affected directly and indirectly by the Conservancies and wildlife PES schemes assessed. In addition, the correlation between areas of high poverty rates and high wildlife biomass suggests that there is potential for PES to impact on poverty. It estimated that, without taking into account the foregone opportunity costs, an annual PES transfer of US$ 10-15 per hectare would be sufficient to close the poverty gap in much of the ASALs. It concluded that PES funding can currently only be realized with financial support from the public sector especially through government funding. It further noted that climatic variability recorded showed increases in temperature and declines in long season rainfall in the period between 1960 and 2009. These trends are expected to continue in the short term to 2025, altering climate pattern and affecting pastoral livelihoods through increased drought and flooding events. This necessitates the need for strengthening coping and adaptive strategies among pastoral communities living in Kenyan ASALs. Chapter 4 addresses the issue of climatic variability, specifically drought and the potential role of PES as a coping and risk management mechanism within the context of ecosystem based adaptation to climate change. The chapter includes analyses of drought at the local level based on specific case studies of PES schemes that are currently being implemented among the Maasai pastoralists at the two study sites.

CHAPTER 4: EXPLORING THE ROLE OF PAYMENTS FOR ENVIRONMENTAL SERVICES IN ECOSYSTEM BASED ADAPTATION TO CLIMATE CHANGE AND POVERTY ALLEVIATION: INSIGHTS FROM KENYAN RANGELANDS

Philip Osano and Joseph O. Ogutu

Abstract

This paper examines the role of payments for environmental services (PES) in ecosystem based adaptation to climate change and poverty among pastoral communities in Kenyan rangelands. It first presents a conceptual framework of the inter-linkages between PES and ecosystem-based adaption (EBA) followed by an analysis of drought occurrences from 1914 to 2011 in the Maasai Mara Ecosystem (MME) and from 1960to 2011 in the Athi-Kaputie Plains (AKP). It also includes an assessment of the effects of PES on the adaptation of pastoral PES participants to climate change, and on the local institutions relevant to climate change adaptation. Locally, droughts have increased in frequency and severity at the two sites. The PES effects on three determinants of adaptation are identified. First is on economic assets and wealth where PES is seen as a risk management diversification strategy to a relatively stable and predictable income source in the short term. It may serve as a 'safety-net' to pastoral families in the drought periods. Second, PES can enhance human capital, access to technology and infrastructure directly through training and educational programs and by providing access to technology and infrastructure to improve basic coping strategies such as water and hay storage, or indirectly if PES income is invested in education and health. Third, PES can influence how local landowners are engaged in the local governance and in decision making concerning land use planning. Lastly, four main effects on institutions relevant to adaptation are identified: PES establishes new rules and regulations that influence patterns of pastoral land use and management; it leads to the creation of collective action institutions which shape communities local adaptation responses and basic coping strategies including mobility, storage, diversification, communal pooling, and market exchange; and it creates inter-sectoral and cross-sectoral linkages. There is

however some concerns with regard to the equity, leakages and financial sustainability of the PES schemes assessed.

Introduction

The provision of environmental (or ecosystem) services (ES) for human well-being is a high priority in environment and development policy across the developing world (TEEB, 2009), and particularly in dryland ecosystems which have experienced very high rates of ecosystem degradation (Safriel and Adeel, 2005). The ecosystem services framework;"the direct and indirect contributions of ecosystems to human well-being" (TEEB, 2010) provides a basis to relate biodiversity and ecosystem conservation to economic development, including poverty reduction (Tallis et al., 2008). Many ecosystem services are generated in dryland ecosystems (Safriel and Adeel, 2005), which provide a critical source of food, income, employment, nutrients and risk insurance to local communities through the extensive livestock production (Perry and Sones, 2007).

Currently, several ecological and socio-economic processes of change are occurring in the dryland systems, with implications for rangelands and generating effects on biodiversity and traditional pastoral livestock production (Hobbs et al., 2008b). Two most critical issues are climate change and increase in human poverty. Climate change could lead to negative impacts on the livelihoods of pastoral communities and is becoming a critical concern in rangelands (McCarthy et al., 2001). This is because the *adaptive capacity* of pastoral rangeland communities is eroding making them become more vulnerable to climate-induced and other environmental changes (Fraser et al., 2011, Western and Manzolillo Nightingale, 2003). Building the adaptive capacity of pastoralists in Africa has thus become a major priority in humanitarian, development and environmental management communities (Mude et al., 2007, Nori and Davies, 2007, Hurst et al., 2012)

In the context and processes of climate change adaptation, which is defined as "the adjustments in ecological, social or economic systems in response to actual or expected climate stimuli and their effects or impacts" (Smit and Pilifosova, 2001), the concepts of *vulnerability*, *adaptive capacity* and *resilience* are closely linked (Janssen and

Ostrom, 2006). The *vulnerability* of a given system or society is considered as "a function of its physical exposure to the effects of climate change and its ability to adapt to these conditions" while the *adaptive capacity* refers "to the potential, capability or ability of a system to adapt to climate change stimuli or their effects or impacts" (Smit and Pilifosova, 2001). Both the *vulnerability* and *adaptive capacity* are closely associated with poverty (Eriksen and O'Brien, 2007). In pastoral communities, the management or governance over natural resource use is considered as the core of pastoral adaptive capacity, and it includes formal and informal institutions (Galvin, 2009). *Resilience* refers to the amount of change a system can undergo and still retain the same function and structure while maintaining options to develop and is highly desirable for enhancing the adaptive capacity of communities (Nelson et al., 2007).

Studies have shown that climate change will disproportionately affect livestock systems in the developing world (Thornton et al., 2009), and these effects will be more pronounced among pastoral communities in African rangelands (Thornton et al., 2006, Ericksen et al., 2013). The effects of climate change are expected to further complicate the livelihoods of African pastoralists, already facing threats to their traditional lifestyle that arise from population growth; loss of herding lands to private farms, parks and urbanization; transformation of land ownership (tenure) from common to private property leading to sedenterization; and periodic dislocations brought about by drought, famine and civil war (Fratkin, 1997, Galaty, 1994b, Galaty, 1994a). Taken together, these threats increase the vulnerability of pastoralists in Africa to the effects of climate change, leading to further poverty and increased potential for conflicts with other pastoral communities and farmers (CDC/IISD/Saferworld, 2009, Eriksen and Lind, 2009). The end result may be a spiral process where poverty contributes to increased vulnerability to climate change, which in turn leads to outcomes that further perpetuate poverty (Eriksen and O'Brien, 2007).

Sustainable adaptation is proposed as a means of addressing both climate change vulnerability and poverty among pastoral communities (Eriksen and O'Brien, 2007), but while sustainable adaptation is considered a prerequisite for sustainable development in rangelands (UNCCD/UNDP/UNEP, 2009), there is currently little understanding of both the sensitivity of pastoral livelihood strategies' and the adaptive capacity to deal with climate change in the long term (Nassef et al., 2009). In the arid and semi- arid (ASAL) areas in the East and Horn of Africa, pastoralists are diversifying their livelihoods as a result of widespread and sustained poverty, in response to changing global markets, new technologies, new crop and livestock breeds (Catley et al., 2013), and also as a response to the processes of land privatization, subdivision and sedenterization (Aboud et al., 2012b, Homewood et al., 2009c). Wildlife PES is one of the tools of livelihood and income diversification into tourism that is expanding among pastoral landowners living close to wildlife protected areas (Bulte et al., 2008a, Sachedina and Nelson, 2012).

This chapter looks at the PES schemes in which direct payments are provided to landowners for biodiversity conservation (mainly wildlife conservation). Under these PES schemes, pastoral land users are compensated or rewarded with payments derived from public funds (mainly state wildlife and protected area agencies and non-governmental sources) and/or private sources (mainly commercial tourism operators) for their stewardship of landscapes or wildlife that have scenic or recreational values to tourists, and for maintaining certain land uses that are compatible with wildlife conservation.

Although PES is mainly aimed at supporting the provision of ecosystem services, it is claimed that in the rural areas of the developing world, PES can also generate co-benefits such as poverty reduction and food security (FAO, 2007, FAO, 2011) and contribute towards ecosystem based adaptation to climate change (van de Sand, 2012, Wertz-Kanounnikoff et al., 2011). Little attention is paid however to the cost incurred by the different rural groups in rural areas such as youth, women and the landless of PES implementation. These costs may include land appropriation, loss of access to natural resources that provide food and medicinal material for rural communities and loss of livestock grazing lands through land use restrictions imposed through PES regulations.

In terms of climate change, so far the focus of PES has been to promote climate mitigation (for example, carbon sequestration through payments to land users to plant or maintain trees to sequester carbon in forests and mixed crop-livestock systems) but not adaptation to climate change (van de Sand, 2012). In addition, very few PES schemes are currently operational in rangelands in general, and the few that exists are mostly in the developed countries such as the United States (Goldstein et al., 2011) and Australia (Greiner et al., 2009). There is, however, a potential for PES in the rangelands of the developing world. Already some pilot PES schemes are have been developed in north Africa and in different countries in sub-Saharan Africa (Duttilly-Diane et al., 2007, Dougill et al., 2012, Perez et al., 2007, Tschakert, 2007).

Some authors have argued that implementation of PES in rangelands could yield a triple win by improving the livelihoods of pastoral communities, mitigating climate change and contributing to biodiversity conservation (Neely and Bunning, 2008, Silvestri et al., 2012). There is however little evidence to support the triple-win argument, let alone, win-win outcomes.

With respect to PES and climate change adaptation, it is hypothesised that PES can potentially help participating land users to adapt to climate change but could also generate risks that can undermine such adaptation efforts (van de Sand, 2012). Among pastoralists, the implications of PES on climate change adaptation remains unclear (Silvestri et al., 2012, Birner and Osano, 2012).

Natural resources, including biodiversity and ecosystem services play a critical role in the adaptive capacity of pastoral communities, in particular, by increasing pastoral resilience, and reducing their vulnerability to climatic variability and climate change (Mortimore et al., 2009, Nori and Davies, 2007). Some studies have shown that PES can support ecosystem-based adaptation (EBA) to climate change in natural resource-dependent societies (Wertz-Kanounnikoff et al., 2011, World Bank, 2009). EBA is defined broadly as; "the use of biodiversity and ecosystem services as part of an overall adaptation strategy to help people adapt to the adverse effects of climate change" (Secretariat of the Convention on Biological Diversity, 2009).

In the context of ASALs, the sustainable management of grasslands and rangelands to enhance pastoral livelihoods and the conservation of wildlife habitats is presented as a form of EBA with the potential to generate multiple socio-cultural (recreation and tourism), economic (income for local communities), and biodiversity (forage for grazing animals and wildlife habitats) co-benefits (Secretariat of the Convention on Biological Diversity, 2009). In reality, the majority of pastoral and agro-pastoral communities have no experience with PES as only few PES schemes exists in the ASALs. As a result, there is a lack of information about the implications of PES for climate change adaptation among pastoral communities, including the potential of PES as a mechanism for EBA. Policy makers have identified this as a key research priority issue (Government of Kenya, 2010).

Climate change is a long term phenomenon, but in the short-term, its' effects are manifested in terms of the variability in precipitation and temperature (Funk et al., 2010). This chapter considers climate change adaptation in the short-term, looking at the implications of PES on pastoralists' drought coping and risk-management strategies. Although pastoral communities have, over long periods of time, developed and evolved indigenous ways of adapting to drought, climate change has brought new challenges that make these indigenous adaptation strategies inadequate (Nassef et al., 2009). A recent report on climate change in African dryland systems suggests that the future of pastoralism will, to a large degree, depend on the ability of pastoral communities to pursue adaptive strategies in three key areas; the management of natural resources, the expansion of market opportunities, and the strengthening of local institutions (UNCCD/UNDP/UNEP, 2009).

An innovative approach to the management of dryland natural resources in Africa is the reward and compensation to pastoral land users for the ecosystem services provision resulting from their land use and management practices such as the provision of habitat for wildlife and the carbon sequestration (Bulte et al., 2008a, Dougill et al., 2012). A large share comprising 90% of pastoralists' income is derived from livestock (Upton, 2004), but this income source is unstable and fluctuates seasonally, recording large declines during drought period because of price and market distortions (Barrett et al., 2003). As an income diversification option that is regular, stable and relatively predictable, PES can potentially stabilise pastoralist's income thereby also serving as a drought coping and risk mitigation mechanism.

A prerequisite for PES implementation is the identification of the ecosystem service users/buyer willing to pay for the ecosystem services or the land use that supports the provision of the desired ecosystem services (Engel et al., 2008). In Kenya, the law regulating wildlife management prohibits the consumptive utilisation of wildlife such as trophy hunting which has been a successful way of creating demand for wildlife services in southern Africa (Naidoo et al., 2011, Frost and Bond, 2008). The demand for wildlife services is therefore restricted to non-consumptive uses such as wildlife tourism, education and nature preservation through protected areas (Norton-Griffiths, 1998).

The demand for wildlife services comes therefore mainly from the direct non-consumptive users that comprise largely of three groups; the state agencies that are responsible for wildlife and forest conservation and for the management of protected areas such as the Kenya Wildlife Services (KWS) and the Kenya Forest Services (KFS); the conservation groups and NGOs that are interested in species and habitat conservation, and the private sector companies in the tourism industry interested in nature-based tourism.

The establishment of PES also requires that land tenure rights are clarified (Vatn, 2010). This process is difficult and complicated in pastoral rangelands which are managed under a mix of traditional and modern land tenure and where land and resources rights are tenous (Unruh, 2008). In Kenyan rangelands, there is an ongoing process of land privatisation and sub-division shifting land tenure from communal to private ownership to provide individual pastoral landowners with the full rights of control to their land (Galaty, 1994a).

Following land adjudication and sub-division, pastoralists obtain title deeds which they are able to use to enroll their land in PES schemes (see Chapter 5 and 6). Furthermore, the title deeds provide landowners with tenure security which enables them to capture wildlife rents, including from tourism and PES, directly at the household level (Norton-Griffiths and Said, 2010), as opposed to through community level institutions, where such benefits are generally mismanaged and distributed among a few elites .

Much attention concerning the livelihoods implications of PES in livestock production systems has focussed on the direct benefits of income provision and poverty reduction but little attention has been paid to the indirect benefits/co-benefits of PES, such as its role in climate change adaptation (Silvestri et al., 2012). Even worse, very little or no attention has been directed to the costs of PES, including the differential impacts at intra-household level , including the fact that livestock and livestock products (meat, milk, hide etc) deliver benefits to all household members (men, women, and children). Wildlife PES however, preclude these benefits and instead delivers a cash payment to the head of the household, usually a man, a benefit which may or may not be shared equitably within the household.

PES and Ecosystem-based adaptation strategies have largely been pursued in isolation, and where there have been attempts to link these two strategies, it has mostly been conceptual rather than empirical (van de Sand, 2012, Wertz-Kanounnikoff et al., 2011). Furthermore, despite the existence of a rich literature on traditional drought management strategies among pastoral communities, no attempts have been made to situate PES in the context of drought risk management and long term climate change adaptation.

This chapter seeks to fill this knowledge gap by examining how wildlife PES programs affect pastoralists' drought coping and risk mitigation strategies in the short term and their potential role in promoting ecosystem based adaptation to climate change in the long term. It brings an empirical perspective to this debate by drawing on descriptive case studies of two PES schemes among pastoralists in southern Kenyan rangelands to provide insights for informing policy and future research in this emerging area of enquiry.

The chapter responds to the following three questions:

1) What is the trend in drought occurrence in the two study sites based on recorded frequency and severity?
2) What are the effects of PES on the adaptive capacity of pastoral households as environmental service providers?
3) What are the effects of PES on the local level institutions relevant to climate change adaptation in the two study sites?

The chapter first provides an overview of pastoralism, wildlife tourism and poverty in relation to climate change in Kenyan ASALs. It then proposes a conceptual framework linking PES to ecosystem based adaptation (EBA) to climate change among pastoral communities. These are then followed by the methods, results, discussion and the conclusion.

Pastoralism, wildlife tourism and poverty in relation to climate change in Kenya

More than 80 percent of Kenya's land is classified as ASALs, which are characterized by low, unreliable and variable rainfall. In these areas, extensive livestock production, in the form of pastoralism and wildlife conservation, are considered as the most suitable land uses. Recent estimates shows that ASALs contain 70% of cattle, 87% of sheep and 91% of goats found in Kenya (Behnke and Muthami, 2011).

The importance of livestock in ASALs is underscored by the fact that it is the main source of economic security for pastoralists and accounts for more than 95 percent of family incomes (FAO, 2005b). Although it is difficult to estimate the exact off-take rate of cattle in the ASALs as this is subject to weather variability and market prices, recent studies provide an off-take rate estimate of 4.4% per annum between 1970s and 1990s (Norton-Griffiths and Said, 2010). It is estimated that in 2009 alone, the ASALs recorded a net off-take of cattle for sale and slaughter of 1.83 million heads of cattle (Behnke and Muthami, 2011). These figures are projected to rise in tandem with the increasing market demand for livestock due to population growth, increased incomes and urbanization (Omiti and Irungu, 2002).

Although this suggest a huge potential for growth in pastoral livestock production which could benefit pastoralists economically, this potential has been hampered by policy constraints, including a lack of institutional support, which has stifled the ability of pastoral communities to boost their economies through livestock sales. Consequently, the majority of pastoral communities have low incomes, which is made worse by their limited access to infrastructure and essential services such as education, health, water and sanitation, resulting in high poverty rates in ASALs (Republic of Kenya, 2012b).

The ASALs are also the cornerstone of Kenya’s wildlife sector and tourism industry. An estimated 88 percent of protected area land and more than 70 percent of the country’s wildlife resources, including the larger wildlife and mega-herbivores that are dispersed either permanently or seasonally outside the protected areas in private and communal pastoral lands, are in the ASALs (Norton-Griffiths and Said, 2010). This rich diversity of wildlife in the ASALs makes an important contribution to the prosperity of Kenya’s tourism industry which is a critical sector of the Kenyan economy (Dieke, 1991, World Bank, 2011).

The majority of the large mammals use protected areas for dry-season grazing and move out to the wet-season dispersal areas located on pastoral lands. As a result of these migrations, cases of wildlife coming in direct conflict with farmers and pastoralists in private and communally owned lands are common (Okello, 2005, Sindiga, 1995) and these increase in frequency and severity during drought periods (CDC/IISD/Saferworld, 2009). As a consequence of both the protected lands allocated to wildlife and the conflicts between mega-fauna and local populations, pastoral communities (as landowners and mobile livestock keepers) incur high costs of wildlife especially near major wildlife protected areas (Sindiga, 1995).

Although wildlife tourism in Kenya generates billions of dollars annually (World Bank, 2011), pastoral communities are not adequately compensated for the associated costs and little tourism revenue trickles down to the pastoral landowners whose land supports wildlife outside the protected areas (Kabiri, 2010). Additionally, existing land tenure and property rights in Kenya do not provide sufficient economic incentive for landowners to keep wildlife on their private lands (Kameri-Mbote, 2005, Norton-Griffiths, 1996). Pastoral landowners contribute to wildlife losses actively through killing of wildlife predators in direct retaliation for livestock losses, and passively through their adoption of land uses, such as crop cultivation that are incompatible with wildlife conservation.

Climate change will add to the current challenges facing ASAL pastoral populations, pastoral livestock production and wildlife management (Barnes et al., 2012, Kabubo-Mariara, 2009, Kaeslin et al., 2012, Ogutu et al., 2007). Studies show an increasing trend in temperature which is predicted to continue in the long term. Precipitation records show a declining trend in some parts of Kenyan ASAL, which is predicted to continue (Funk et al., 2010), however, there remains considerable uncertainty regarding the long term future precipitation trends;(Williams and Funk, 2010).

Climate change will specifically affect pastoralism through changes in vegetation, frequency of drought and livelihood transitions (Ericksen et al., 2013). Drought effects in particular are becoming a major factor having intensified in ASALs in recent times, increasing in both frequency and severity (Orindi et al., 2007).

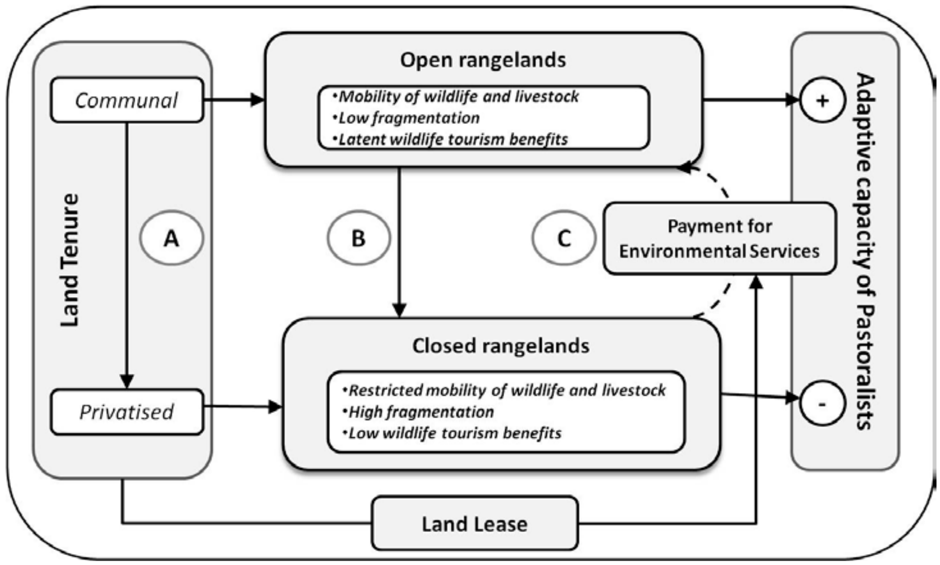
Severe droughts are marked by extreme scarcity of water leading to large declines in biomass production (grass and shrubs) and reduce fodder availability and resulting in high mortality of livestock, especially under conditions of constrained mobility of livestock herders (Nkedianye et al., 2011).

Livestock mortality that results from drought also reduces the per capita livestock holdings among pastoral communities to thresholds below which they are unable to recover (Western and Manzolillo Nightingale, 2003). This also generates risks of pastoral households falling into poverty traps especially given the short restocking period in between successive drought events. High livestock mortality can result in increased poverty attributed to loss of income from livestock assets. This in turn can make pastoral families more vulnerable to climatic shocks such as droughts, and in the long term, may reduce their ability to adapt to climate change (Nassef et al., 2009). The end result could be a positive feedback loop of increasing vulnerability to climate change and worsening poverty conditions (Eriksen and O'Brien, 2007).

A conceptual framework linking PES to EBA in Kenyan ASALs

Figure 4.1 is a conceptual model linking PES, in the context of wildlife conservation and tourism in Kenyan rangelands, to ecosystem based adaptation. The framework considers the social-ecological systems of rangelands as having two stable states. The open rangelands is characterised by mobility of livestock and wildlife with low levels of fragmentation and latent wildlife tourism benefits. The alternative, under a closed rangeland condition is characterised by a state with limited mobility options for both livestock and wildlife, high levels of fragmentation and low wildlife tourism benefits.

Figure 4.1 Conceptual framework of PES and ecosystem based-adaptation to climate change in Kenyan rangelands.



The framework can be explained as follows:

- *A: Land tenure* - Transitions between the open and closed rangeland states are driven by changes in land tenure policies. This occur through privatization and sub-division of rangelands from communally owned to individually owned land units (Galaty, 1994a, Kimani and Pickard, 1998);
- *B: Rangeland states* - As rangelands shift from open to closed states, there is a concurrent shift to land uses that constrain pastoral livestock and wildlife mobility (Reid et al., 2008, Reid et al., 2004). The closed rangeland state is characterized by a high vulnerability and low adaptive capacity of pastoralists to climatic variability (Galvin et al., 2004);
- *C: Payment for environmental services* - PES practices modify the management of land by pastoralists and land management encompasses the core of pastoral adaptive capacity (Galvin, 2009). Step C therefore illustrates how PES may influence the adaptive capacity of pastoral households as environmental service providers, and responds directly to the question 2 of this chapter. As an explanation, direct payments based on the lease of private and communal land for wildlife

conservation and tourism may foster post-privatization reversion to open rangelands by restricting land uses that impede mobility of wildlife and livestock across the landscape (Victurine and Curtin, 2010). Income derived from PES can also contribute to poverty reduction among participating pastoral households (Bulte et al., 2008a).

Methods

Study sites and PES schemes

This paper is based on a study carried out between 2009 and 2011 in two sites in southern Kenya. Study site 1 in the Athi-Kaputie Plains, is a dispersal area for the Nairobi National Park used by large herbivores such as zebra and wildebeest in the wet season for feeding and calving; however, changes in land use, including urbanization, expansion of crop cultivation and permanent settlements with fences, have resulted in habitat fragmentation disrupting traditional mobility of pastoral livestock and wildlife (Reid et al., 2008). A PES scheme, the Wildlife Lease Program (WLP), was initiated in 2000 involving pastoral Maasai landowners in the area, whereby participating households are paid US$10 per hectare per year in return for keeping their land open to wildlife, while still allowing them to use the land to graze their livestock (Gichohi, 2003).

Study Site 2, the Maasai Mara Ecosystem, borders the Maasai Mara National Reserve (MMNR) to the north and includes within it the Olare Orok Conservancy (OOC). The OOC is characterized by open pastoral and agro-pastoral lands that contain resident wildlife and also serve as a dry-season dispersal area for wildlife herds such as wildebeest and zebras. Changes in land use, combined with an increase in human population and settlements, land privatization and cropland expansion have isolated land previously used by wildlife (Lamprey and Reid, 2004), and contributed to declines in wildlife populations outside the Reserve (Ogutu et al., 2011). The OOC was started in 2006 after the land sub-division in the former Koiyaki Group Ranch. It involved a partnership between 157 pastoral land owners and four tourist operators. Participating households are required to relocate their settlements away from the conservancy and livestock grazing is also regulated. The tourist operators guarantee fixed payment directly to participating households. The payment ranged from US$20.ha^{-1}.yr^{-1} in 2006 to US$39.ha^{-1}.yr^{-1} in 2009. While mostly similar, the two PES schemes differ in some characteristics, as shown in Table 4.1 below.

Table 4.1 Characteristics of the Wildlife Lease Program (WLP) and the Olare Orok Conservancy (OOC) PES programs
Source; Summary prepared by the lead author.

			Athi-Kaputie Plains	**Maasai Mara Ecosystem**
PES program			**The Wildlife Lease Program (WLP)**	**Olare Orok Conservancy**
Year Scheme Established			2000	2006
PES Actors	ES sellers (participants)	Number of households enrolled (per year)	18 (2000) and 387 (2010)	156 (2006) 277 (2011)
		Total land area enrolled (per year), hectares	688 (2000) 16,774 (2010)	10,040 (2006)
		Average land enrolled in 2010 hectares/HH(range)	102.68 (6.07–971.28)	56.14 (16.19–66.78)
	Intermediary Institutions[1]		The Wildlife Foundation (TWF) (CSI)	OOC Ltd. (CSI) and Ol-Purkel Ltd (PMI)
	Direct ES users/buyers[1]		Kenya Wildlife Service (KWS) (PSI) Global Environment Facility (GEF) (PSI) The Nature Conservancy (TNC) (CSI)	Porini Camp (PMI) Mara Plains (PMI) Kicheche Camp (PMI) Olare Camp (PMI) Virgin Camp (PMI)
PES Design and Implementation	Selection of ES providers		First-come basis, later based on proximity to Nairobi National Park, connectivity to unfenced plots and other leased parcels.	Land parcels in the geographic area designated for use as wildlife and tourism in OOC
	PES contract duration (period)		1 year, renewable (2000–current)	1 year (2006) 5 years (2007) 5 or 15 years (2011–current)
	PES payment features	Amount paid in US$/ha/yr(year)	10 (year 2000– 2010)	20 (2006) ; 33 (2008); 39 (2009)
		Timing of payment	Three payments per year in January, May and September	Payments made monthly
		Payment differentiation (spatial, other)	Uniform payment for all participating landowners	Initially uniform differentiated since 2011 by the period of contract (5 or 15 years)
	Land use restrictions	Restrictions on land use and other activities	No fencing No land subdivision and sales No wildlife poaching	No settlements on OOC land Controlled and managed livestock grazing inside the conservancy

[1]The intermediary and direct buyer institutions are classified into three generic categories: **CSI (**civil society institution); **PMI (**private market institution); **PSI (**public/state institution).

Data and analysis

The primary and secondary data used was collected from various sources (see chapter 3, 4 and 5 for additional details) and analysed using multiple methods. The data on monthly rainfall at the two study sites were obtained from the Kenya Department of Meteorology and was used to calculate the intensity and frequency of drought occurrences for the period from 1914 to 2011 for the Maasai Mara Ecosystem and from 1960 to 2011 for the Athi-Kaputiei Plains. The drought intensity was categorized based on the criteria summarised in Table 4.2 (Ogutu et al., 2007).

Table 4.2 Drought classification based on the percentile of the standardised observed rainfall values.
Source: (Ogutu et al., 2007).

Percentile values	**Drought category**
41-75th percentile	Normal
26-40th percentile	Moderate/mild drought
11-25th percentile	Severe drought
0-10th percentile	Extreme drought

The household socio-economic data was based on household surveys carried out at the two study sites in 2009 and 2010 using detailed, semi-structured questionnaires that were fully completed by a total of 131 households in MME and 164 households in AKP. The surveys were administered by locally recruited Maasai enumerators that were trained in survey data collection. A stratified random sample of PES participants and non-participants at the two sites was drawn. The sampling frame for the participants at both sites was based on the PES enrollment register and non-participants were selected opportunistically in the study areas. The household socio-economic data were used to calculate the changes in per capita livestock holdings and in the various sources of household cash income for the 2008 and 2009 period.

To establish pastoral landowner's perception of the role of PES as a drought coping mechanism and risk mitigation strategy, the survey respondents were asked if they were willing to accept higher PES payment rates during periods of drought in return for lower rates during periods of normal precipitation. They were also asked if they attach different values to PES income in dry and wet seasons.

In addition, formal and informal interviews were carried out with stakeholders at the two sites. The stakeholder involved were land owners (both the PES participants and non-participants), PES program managers, PES funders (tourist operators and government agencies), and key informants in the community. The stakeholder groups directly involved in PES implementation were grouped into three categories. The first category consists of the ecosystem services (ES) “users” or “buyers,” This includes groups or organizations benefitting from ES and paying for the coordination and implementation of management activities for ES. The second are the ES; “providers,” which are the stakeholders, holding contractual relationships with the users, either directly or through intermediaries and who commit to implementing management practices that deliver ES on their land. The last category consists of the “intermediaries,” which are the organizations that define the conservation activities to be performed by the providers to ensure delivery of ES and are also responsible for collecting funds derived from the users in order to pay for the providers (Corbera et al., 2007a).

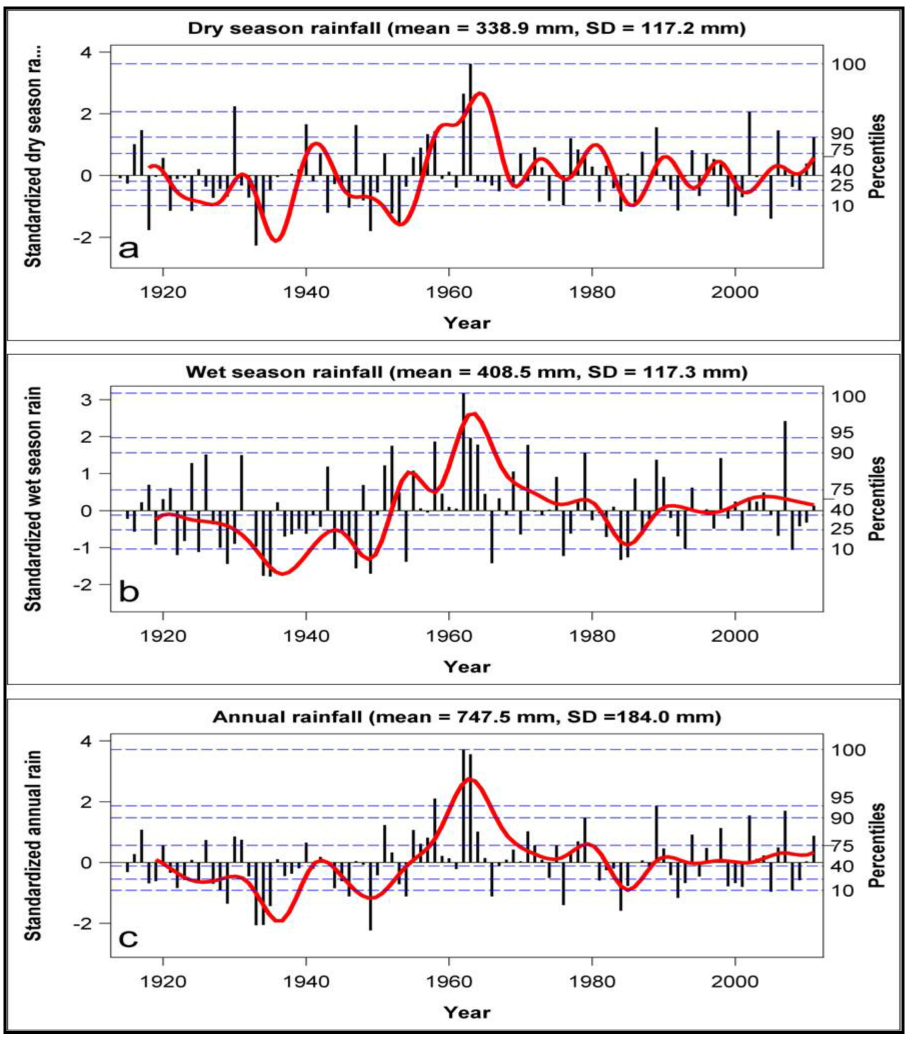
Results and findings

The trends in drought frequency and severity

There were recorded differences in the occurrence, frequency and intensity of droughts in the two study sites but overall, the analysis shows that the extreme and severe droughts are increasing but are localised in the two study sites with the drought in 2008-2009 recorded as severe and moderate in both sites in 2008 and 2009 respectively.

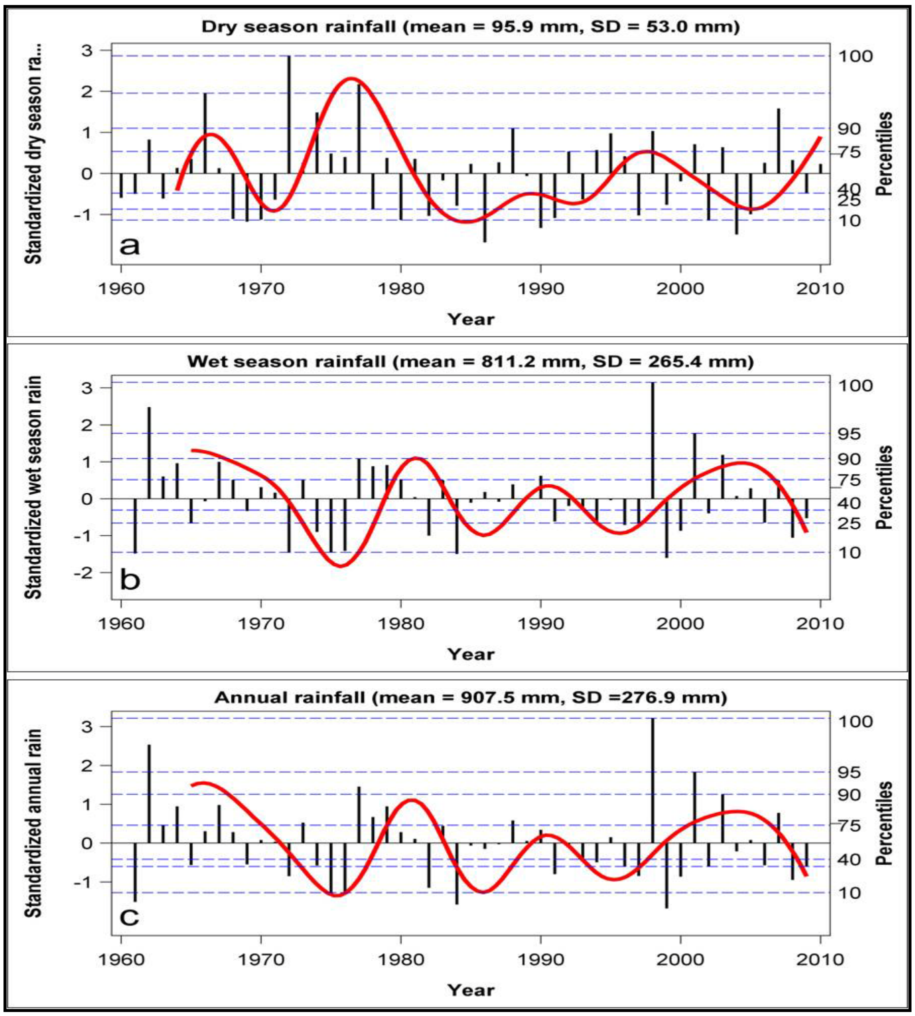
In the MME, droughts in the dry and wet seasons and annually were more frequent and severe in the period 1914-1960 than 1960-2011 (Figure 4.2a, b, c). The analysis of 5-year moving averages shows a quasi-periodic pattern with dry phases often characterized with droughts. Although there is no consistent pattern of increasing or decreasing drought frequency and intensity since 1960, droughts were more common in the 1990s and the 2000s. In this period, a total of four extreme droughts (1966, 1976, 1984, and 1991) and seven severe droughts were recorded (Figure 4.2c).

Figure 4.2 Standard anomalies in rainfall in the MME showing for the period 1914 to 2011, **A**. The dry-season (July-October) rainfall. **B.** The wet-season rainfall (November-June). **C.** The annual (sum of wet and dry-season) rainfall. The dashed horizontal lines are the percentiles. The solid vertical lines are the standardized observed rainfall values. The red solid line indicates the 5-year moving averages.
Data source; Kenya Meteorological Department



In the AKP, there was no clear trend of increase or decrease in the dry and wet season, and the annual rainfall. The long term rainfall records showed a 5-year quasi-cyclical pattern (Figure 4.3a, b, c). A total of five extreme droughts were recorded between 1960 and 2011 (in 1961, 1975, 1976, 1984 and 1999) and a further six severe droughts over the same period (Figure 4.3c).

Figure 4.3 Standardized anomalies in rainfall in the Athi-Kaputiei for the period 1960 to 2011, showing. **A**. The dry-season (June-September) rainfall. **B.** The wet-season rainfall (October-May). **C.** The annual (sum of wet and dry-season) rainfall. The dashed horizontal lines are the percentiles. The solid vertical lines are the standardized observed rainfall values. The red solid line indicates the 5-year moving averages. *Data source*; Kenya Meteorological Department.



The effects of PES on the adaptation of pastoral ES providers

The literature suggests that PES has effects on the *adaptive capacity* of ES providers across four factors: economic assets and wealth; human capital, technology and infrastructure; cognitive factors; and empowerment and local governance (Wertz-Kanounnikoff et al., 2011). Table 4.3 (column 4) lists the effects of the PES schemes on three of these, as identified in the two PES schemes reviewed.

Table 4.3 The factors that determine the *adaptive capacity* of ES providers and the role of PES as applied to the Wildlife Lease Program (WLP) and Olare Orok Conservancy (OOC).
Source: (Wertz-Kanounnikoff et al., 2011) and Lead authors' survey.

Factor	Relevance of factor for adaptation	Scope for PES to affect factor	Applicability to WLP and OOC PES Schemes
Economic assets and wealth	Highly relevant, increases ability to cope with climate change	PES can typically provide small but positive contributions	o PES is a source of pastoral income diversification o PES provides a stable and predictable income but financial sustainability needs to be addressed o PES constitute a high proportion of household income during periods of drought, accounting for 37% and 25% of the total gross income in OOC and the WLP, respectively
Human capital, access to technology and infrastructure	Highly relevant, provides soft skills and engineering solutions to cope with climate change	PES not relevant unless accompanied by training or extension	Directly: o In the WLP, providers trained to use mapping tools (global positioning system [GPS] and geographic information system [GIS]) to map land use in the project area. o In the OOC, a mechanized hay bailing project introduced to improve fodder storage for use and sell during dry season Indirectly: o In the WLP, the highest proportion of PES income (80% in 2009) spent on education need
Empowerment and local governance	Highly relevant to define and decide on sustainable adaptation strategies, information sharing and social learning to cope with climate change	Most user-financed PES schemes have empowered service-providing land stewards and in various cases helped to consolidate land rights	o In the WLP, the ES providers were involved and contributed to the formulation of a land use plan o In the OOC, the ES providers' capacity for negotiation and bargaining was improved, leading to positive adjustments in PES rates and establishment of flexible PES contracts

Economic assets and wealth

The two components of economic asset and wealth considered here are household livestock holdings and cash income. In terms of livestock assets, an analysis of the changes in pastoral household livestock holdings between 2008 and 2009 based on the survey data from the two sites showed overall declines in household livestock wealth. It also showed an increase in the proportion of households having less than 4.5 Tropical Livestock Units (TLU) per capita (Table 4.4.). This is considered as the threshold below which a pastoral household becomes vulnerable to the poverty trap and climate shocks (Lybbert et al., 2004) in the absence of changes to their economic practices.

The data shows that based on average per capita livestock holdings, the MME households were relatively wealthier than those in AKP. There was a recorded decline in livestock wealth between 2008 and 2009 in both sites. The decline was higher in AKP which recorded a drop of 34% points compared to only 5% for MME in the proportion of households below the 4.5 TLU/per capita (Table 4.4).

Table 4.4 Livestock holdings per capita (TLU/Adult equivalent) among Maasai households in the Maasai Mara Ecosystem (n=131) and the Athi-Kaputiei Plains (n=164).
Data source; Lead author's survey

		Household category TLU per capita				**Total**
		< 1	**1 – 1.99**	**2 – 4.5**	**> 4.5**	
	Year	**Maasai Mara Ecosystem**				
Households (%)	2009	5.00	10.00	25.0	60.00	100
	2008	3.00	7.00	25.0	65.00	100
TLU / capita (mean)	2009	0.63	1.55	3.00	13.30	
	2008	0.65	1.31	3.01	13.61	
		Athi-Kaputie Plains				
Households (%)	2009	23.00	23.00	34.00	20.00	100
	2008	7.00	9.00	30.00	54.00	100
TLU / capita (mean)	2009	0.35	1.48	3.14	12.74	
	2008	0.06	1.44	3.25	13.45	

Notes: TLU-Tropical Livestock Unit, a composite index equal to 250 kg of animal weight used to aggregate livestock species with differing weights

A large share of pastoralists' cash income is derived from livestock. An analysis of the changes in pastoral household cash income between 2008 and 2009 based on the survey data from the two sites shows the following findings (Table 4.5). First, the absolute annual gross income of participating households declined overall at both study sites. Secondly, both sites recorded an increase in the absolute PES income, and the share of PES contribution to gross household income. The latter was higher in OOC compared to the WLP in both years. Third, both sites also recorded declines in the absolute livestock income, and in the share of livestock contribution to gross household income. Again, the former was higher in OOC than in WLP in both years, although the latter was higher among participants in the WLP compared to those in the OOC. Lastly, although the OOC recorded an increase in the absolute income from other sources combined (OIC), this declined in the WLP, and both sites recorded increases in the share of contribution of OIC to gross household income (Table 4.5).

Table 4.5 The changes in the annual gross household income and in three income sources (PES, livestock and other income sources combined) of pastoral landowners enrolled in the Olare Orok Conservancy and the Wildlife Lease Program PES schemes between 2008 and 2009.
Data source; Lead author's survey for 2008 and 2009 data, and Nkedianye et al (2007) for 2004 data.

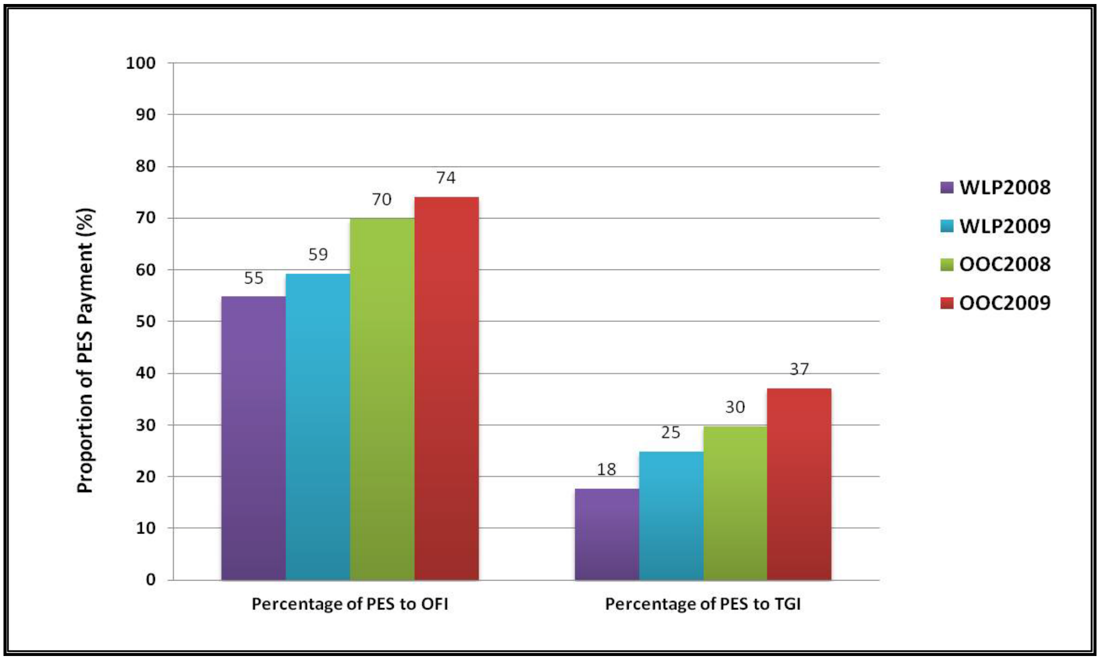
PES Project (Site)	**Year**	**No. of HHs**	**Mean Annual household income (US$/year)**				**Contribution to gross annual household income (%)**		
			Gross	**PES**	**Lives tock**	**OIC**	**PES**	**Lives tock**	**OIC**
OOC (MME)	2008	72	5,315	1,579	2,907	829	30	55	15
	2009	73	5,294	1,963	2,412	919	37	46	17
WLP (AKP)	2004	24		248			7		
	2008	61	1,854	329	1,240	285	18	67	15
	2009	86	1,394	345	800	249	25	57	18

Exchange rate of U$1 = Kenya Shillings (KES) 70.

Table 4.5 shows that although the share of PES contribution to gross household income was less than that of livestock in both sites in 2008 and 2009, it is noteworthy that it was higher than that of all sources of other income combined (OIC) pointing to the relative importance of PES income during a drought period.

This increasing importance of PES in household cash income becomes clear if one looks at the pastoral diversification to off-farm income (OFI) sources. As Figure 4.4 shows, cash income from PES constituted more than half of the share of off-farm income in both the Olare Orok Conservancy and the WLP PES programs in 2008 and 2009. The share of PES in total off-farm income were higher in 2009 than in 2008 in both PES programs, due largely to the effect of the drought.

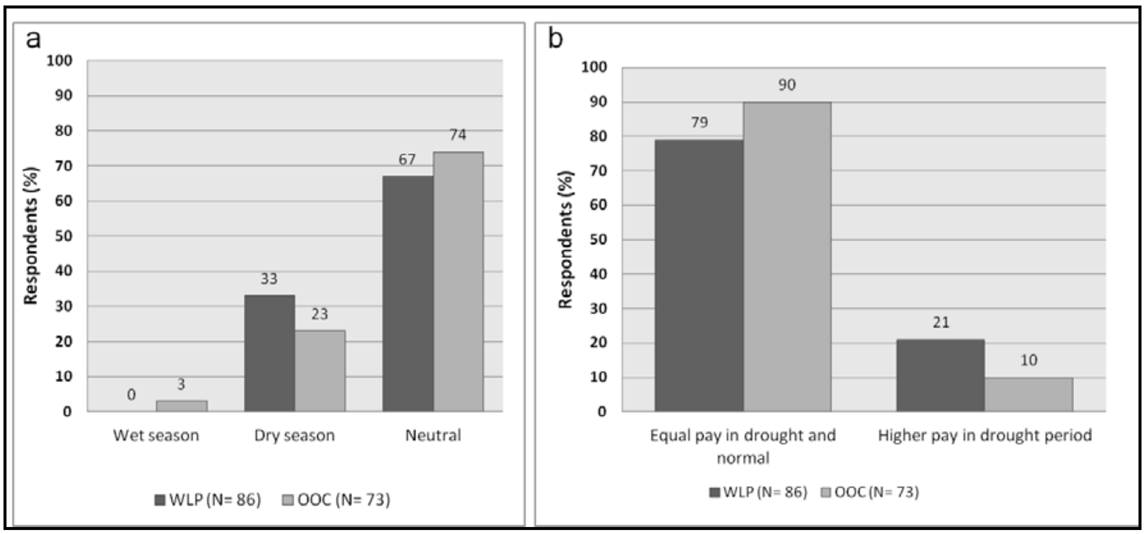
Figure 4.4 The share of annual contribution of PES to household off farm income (OFI) and total gross income (TGI) among participating households in the WLP and the OOC in 2008 and 2009.
Source; Lead author's survey.



The findings from the attitude survey on the potential role of PES in climate change adaptation with respect to drought coping and risk mitigating strategy shows that the majority of the PES participant (67% and 74% of the respondents in the WLP and the OOC respectively) valued PES income equally in both wet and dry season (neutral response). As for the PES participants who reported differences in value to PES income by season, the majority (33% in the WLP and 23% in the OOC) ascribed a higher value for the dry season (Figure 4.5a).

The majority of the PES participants (79% in the WLP and 90% in the OOC) preferred equal PES rates in normal and drought years. Only 21% of respondents in the WLP and 10% in the OOC preferred to receive lower PES rates in normal years in return for higher rates during periods of drought (Figure 4.5b).

Figure 4.5 A. Respondents reporting higher relative value for PES income by season in the WLP and the OOC schemes. **B**. Respondents' reported preference for PES income as a mechanism for coping with risk during drought.
Source; Lead author's survey.

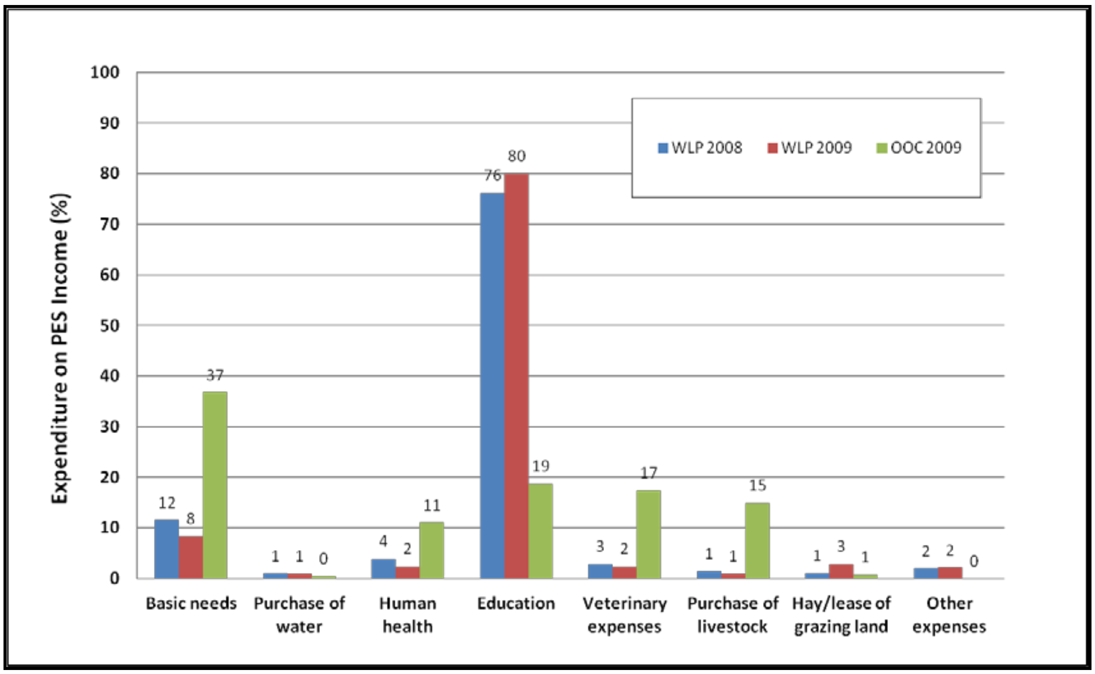


Human capital, access to technology and infrastructure

The implementation of PES can contribute directly and indirectly towards enhancing the *adaptive capacity* of participating households through building human capital, providing access to technologies and relevant infrastructure. The WLP has directly contributed towards enhancement of the human capital as some ES providers received training on land use planning and management. This has involved skills enhancement in the use of participatory mapping technologies such as GPS and GIS to capture information on the location of fences, water sources, roads, towns and open pasture land, and to trace the movement of livestock and wildlife.

In both the WLP and the OOC, there have also been indirect contributions to human capital since participating households have invested their PES income in different forms of capital, including human capital (Figure 4.6). Educational expenses accounted for the highest share of the proportion of expenditures from PES-generated income in the WLP (>75% in 2008 and 2009), and second highest in the OOC (19% in 2009). Participants reported using the money to pay for school fees and purchase of books and school uniforms. In the OOC, the highest proportion expenditures from PES-generated income (37% in 2009) was allocated to basic household needs, such as the purchase of food, clothes and other necessities (Figure 4.6).

Figure 4.6 Respondents' reported PES income expenditures in the WLP (2008 and 2009) and in the OOC (2009).
Source: Lead author's survey.



The indirect contribution of PES to access to technology and infrastructure also involves the investments in storage of water and pasture. In Olare Orok Conservancy, the ecosystem service users represented by the Olare Orok Conservancy Trust (OOCT) which operates alongside the Ol Purkel have in response to the scarcity of water and forage in the study area during the dry season and drought period provided mechanised technologies and infrastructure for the harvesting and storage of water and fodder in form of an *ex-situ* grass bank project. The water is dug from boreholes and distributed through underground pipes to pastoral landowners in the OOC and the local community living in the Talek area. Figure 4.7 shows the Mpuai community water project, which is supported and operated by the OOCT as part of its outreach to the members and the local community, which supports both the households enrolled in the Olare Orok Conservancy as well as those that are not enrolled in the Conservancy.

Figure 4.7 The Mpuai community water project built and being maintained by the Olare Orok Conservancy Trust in the Talek area.

Photo credit: Philip Osano

The shortage of pasture in the dry season and during drought periods brings in challenges for the PES scheme such as in the OOC where livestock grazing inside the Conservancy is restricted. Although traditionally, pastoralists ensured availability of biomass supply for cattle during droughts by setting aside portions of grazing areas pasture or “grass-bank” known as “*Olopololi*” in the local Maa language, this is no longer possible in many parts of the study area due to privatisation and sub-division of land. The lack of dry season pasture refuges creates a challenge for the semi-nomadic communities such as the Maasai who have to move long distances with their livestock herds in search of pasture and water. To address this challenge, the Olare Orok Conservancy through the OOC Trust has set up grass-banks *ex-situ* involving a mechanized hay bailing and storage to provide fodder which can be used and sold to pastoralists during the dry season or drought period (Figure 4.8).

Figure 4.8 A photo of an off-site grass bank in the Olare Orok Conservancy showing cattle grazing in the foreground.

Photo credit: Philip Osano

Empowerment and local governance

The involvement of pastoral landowners in both the WLP and OOC PES schemes has clearly changed the dynamics of pastoral engagement in local and regional governance processes both in terms of empowerment of sections of the community, but also disempowerment to others. In some cases such as in the WLP in Athi-Kaputiei, it is evident that a large section of pastoral landowners have enhanced their participation in decision making about the governance and management of their land. This is evident through the involvement of the WLP PES participants and other local pastoral landowners in the formulation and development of a land use plan for the Athi-Kaputiei Plains, the first of its kind for a pastoral area in Kenya. In the OOC, the capacity of providers to negotiate with the users (tourist enterprise investors) through the *OOC Landowners Co. Ltd*, which is the land owners company has gradually improved, for example, by enabling the conservancy member to bargain for higher PES rates and a flexible contract option that provide for a five or a 15 year contract options. In contrast, these schemes have also disempowered sections of the local communities, in some cases leading to local level conflicts among the community members themselves. In the OOC for example, infractions of livestock belonging to non-members into the conservancy land which they hitherto would have accessed prior to land sub-division and conservancy establishment is a source of conflict between PES participants and other households not enrolled in the conservancy.

Effects of PES on local institutions relevant to adaptation

Institutions are one of the determinants of communities' adaptive capacity to climate change (Smit and Pilifosova, 2001). PES can alter the institutional preconditions for adaptation in three main ways through; local institutions, intra-sectoral linkages and cross-scale linkages (Wertz-Kanounnikoff et al., 2011). In the first case, a review of the PES schemes shows that new local institutions have been created to support or augment the implementation of PES at the two study sites. Two institutions have been established as part of the implementation of the WLP. These are *The Wildlife Foundation* (TWF) which is a non-governmental organization (NGO), founded purposely to act as a PES intermediary, and the *Kitengela I'l Parakuo Land Owners Association* (KILA), which is a local association of pastoral landowners that serves as a forum for collective action to promote the interest of the membership concerning land-use issues in Kitengela which is the northern portion of the AKP.

Three local institutions have been established in the Olare Orok Conservancy. These are the *OOC Landowners Co. Ltd*., established by the pastoral landowners as a not-for-profit shareholding company, and the *Ol Purkel Ltd*., a company limited in liability formed by the users (tourist companies that lease land) to run the Conservancy operations. Both companies represent collective action institutions and also serve as intermediaries between the ES providers and buyers. In addition, the *Olare Orok Conservancy Trust* (OOCT) was established as a mechanism to channel donor funding to support community projects that are not directly part of the Conservancy contracts. Through their activities, these new institutions influence positively and/or negatively these pastoral communities' drought-coping strategies including mobility, storage, diversification, communal pooling and market exchange.

The other two ways in which PES alters the institutional conditions for adaptation is through inter-sectoral and cross-sectoral linkages. The implementation of PES in both sites has involved multiple sectors, thereby organically creating these linkages that involve the institutions and sectors listed in Table 4.6 and 4.7 respectively. The institutions involved in PES represent sectors such as land (rangeland land use and management), tourism, wildlife, environment, and local and central government administrations. Third, the institutions involved in PES implementation also operate at different scales of governance (local, national and international), creating cross-scale linkages (Table 4.6 and 4.7).

Table 4.6 Cross-scale and inter-sectoral linkages involving pastoral ES providers in the Wildlife Lease Program.
Source: Lead author's survey.
Acronyms: **TWF** – The Wildlife Foundation; **KWS**: Kenya Wildlife Services; **GEF**: Global Environment Facility; **KILA**: Kitengela IL Parakuo Land Owners Association; **TNC**: The Nature Conservancy; **IFAW**: International Fund for Animal Welfare; **ILRI**: International Livestock Research Institute; **NGO** – Non Governmental Organisation; **CBO** – Community Based Organisation.

Organization/Institution	**Type of Organization/ institution**	**Sector**	**Level of social organization**			
			Community/ local level	County	National	International
TWF	Local NGO	Wildlife			X	
KWS*	Government institution	Wildlife			X	
GEF/World Bank*	Financial institution	Finance			X	X
TNC*	International NGO	Biodiversity				X
Wildlife Trust	International NGO	Wildlife				
Friends of Nairobi National Park	Membership organization	Wildlife	X			
KILA	CBO	Community	X			
Department of Physical Planning (Ministry of Lands)	Government institution	Land		X	X	
Ol Kejuado County Council	Local government	Governance		X		
Africa Wildlife Foundation	Regional NGO	Wildlife				X
Africa Conservation Centre	Regional NGO	Biodiversity				X
IFAW	International NGO	Wildlife				X
ILRI	Research institute	Research				X

*PES user or buyer

Table 4.7 Cross-scale and inter-sectoral linkages involving pastoral ES providers in the OOC.
Source: Lead author's survey.
Acronyms: **OOC**: Olare Orok Conservancy; **NEMA**: National Environment Management Authority; **KWS:** Kenya Wildlife Service; **IFAW**: International Fund for Animal Welfare; **MoTW**: Ministry of Tourism & Wildlife.

Organization/Institution	**Type of Organization/ institution**	**Sector**	**Level of social organization**			
			Community/local level	County	National	International
OOC Ltd.	Limited company	Wildlife	X		X	
Ol Purkel Ltd.	Limited company	Management	X			
Porini Camp (Gamewatchers Safari)*	Private company	Tourism				X
Mara Plains (Great Plains Conservation)*	Private company	Tourism				X
Olare Camp*	Private company	Tourism				
Kicheche Camp*	Private company	Tourism			X	
Rekero Camp*	Private company	Tourism			X	
Virgin Camp (Virgin Atlantic)*	Private company	Tourism				X
Olare Orok Trust		Development	X			
Koiyaki Guiding School	School	Tourism		X		
Tusk Trust						X
Bill Winter Safaris	Private company	Tourism				X
Howard Saunders: Ker and Downey Safaris	Private company	Tourism				X
IFAW	International NGO	Wildlife				X
Anne Kent Taylor Foundation	Foundation					X
The Lakeside Foundation	Foundation					X
Narok County Council	Government institution	Governance	X			
NEMA	Government institution	Environment			X	
MoTW	Government institution	Tourism/Wildlife				
KWS	Government institution	Wildlife				
Other Conservancies in MME**		Tourism/Wildlife				

*PES user or buyer; **Other Conservancies in MME include Motorogi Conservancy, Naboisho Conservancy, Mara North Conservancy, Ol Kinyei Conservancy and Ol Choro Oiorua Conservancy.

Discussion

The implications of droughts and the role of PES in pastoral adaptive capacity

The findings presented in this chapter provides evidence that droughts in the two study sites are become more recurrent with many extreme and severe droughts recorded in the last two decades (1990s and 2000s) than in the three decades earlier (1960s, 70s and 80s). Drought affects the adaptive capacity of pastoral communities in two major ways. First it leads to high livestock mortality which results in the decline in per capita livestock holding among pastoral household. The 2008-2009 drought for example disrupted the livelihoods of the majority of the Maasai pastoralists southern Kenya as majority of them lost up to three-quarters of their cattle as a result (Osano, 2011).

Second, droughts also lead to the reduction in the cash income derived from the sale of livestock and its products creating short-term liquidity constraints for pastoral households. The combination of low per capita livestock holdings and reduced cash income from livestock assets can lead to increased livestock and income poverty and destitution, and make pastoral livelihoods more vulnerable to climatic and other shocks. The income from PES is critical during droughts because it buffers pastoral families from a precipitous fall in livestock income thereby helping them overcome short-term liquidity constraints arising from the effects of droughts (see Chapter 5 and 6).

The findings also show that the droughts at the two sites are localized, necessitating the need for mobility of herders and their livestock between these two sites and to other pastoral areas in southern Kenya and northern Tanzania in search of pasture and water (Nkedianye et al., 2011). Our analysis highlights the fact that PES can play both a positive and a negative role in relation to the adaptive capacity of pastoralist. This role can be positive by enhancing pastoralists adaptive capacity if it promotes land use practices that keep rangelands open to support pastoral mobility, which is an important adaptation strategy for pastoral and agro-pastoral populations in Africa (Niamir-Fuller, 1999). However, this role can be negative if the PES promoted land use in such as in the

OOC restricts livestock grazing, thereby negatively affecting local livestock herders as well as seasonal herders escaping localised drought in AKP and other sites in southern Kenya and northern Tanzania coming into the MME area in search of pasture and water.. During the 2005-2006 drought for example, herders from MME and other parts of southern Kenya and northern Tanzania moved with their livestock to AKP (Nkedianye et al., 2011) and during the drought of 2008-2009, many herders from AKP moved with their livestock to the MME (Philip Osano, personal observation).

Although our analysis has generated some information regarding the role of PES in drought coping and risk mitigation in the short term, it remains unclear what role PES can play in relation to climate change in the long term. More frequent and severe droughts will have considerable negative impact on both livestock and wildlife. The PES schemes reviewed generate funding through wildlife tourism, which can also be constrained by climate change in the long run (Kaeslin et al., 2012).

The integration of drought contingency planning with climate change adaptation strategies in pastoral areas is called for to ensure that mechanisms such as PES that enable pastoralists to cope with short term drought effects are complemented by other long term mechanism for improving pastoral adaptive capacity (Zwaagstra et al., 2010). The policy responses to drought needs to incorporate PES as an EBA strategy, take advantage of local community participation and focus on long term planning that move beyond the traditional humanitarian relief approach (Osano, 2011, Osano, 2012).

The implications of PES on the adaptation of pastoral ES providers

PES can affect the adaptive capacity of providers through its influence on four factors, three of which were considered in this analysis: economic assets and wealth; human capital, access to technology and infrastructure; and empowerment and local governance (Wertz-Kanounnikoff et al., 2011). Economic assets and wealth are key determinants of climate adaptation (Smit and Pilifosova, 2001). Droughts erode the livestock assets of pastoral families because of the high mortality rates of cattle, sheep and goats that result from shortages of pasture and water. This also reduces the cash income obtained from livestock, which comprise a large proportion of the total income of pastoral families. Furthermore, although livestock provides a large share of cash income for pastoralists, it is prone to seasonal fluctuations and high market volatility (Barrett et al., 2003). PES income, on the other hand as is shown in the analysis can be predictable, regular and stable as long as it remains available. Thus PES can serve as a reliable source of income diversification that provides a critical safety net against the loss of livestock income during droughts and also provides a large share of pastoral household's cash income.

Income diversification among pastoral communities in African rangelands has three dimensions: poverty strategies driven by necessity; risk-management strategies making the best of difficult, unpredictably changing ecologies and economies; and strategies of wealth investment and accumulation (Homewood, 2008). The diversification process observed in the PES schemes in both the WLP and OOC are more attuned to risk-management. As land based PES schemes, the participation in these schemes are restricted to landowners, hence the direct benefits excludes the extremely poor families that are landless, which is a factor that calls for further research to understand the full implications of PES on the landless.

Surprisingly, despite the critical role that PES income played in buffering participating households from the declines in livestock income in the 2008-2009 drought, the majority of PES participants perceived equal PES rates in both drought and normal periods as a desirable payment option. At the same time, the majority of PES participants attached the same value to the income from PES in the wet and dry seasons alike. The few that did not preferred higher PES rates in the dry season, rather than in the wet season. Two factors can explain these findings. One is that from the reported expenditure patterns of PES households, the majority use the PES income to pay bills such as basic needs and educational expenses that are constant and not varied by season. It thus makes sense for the PES rates not to be varied by seasons. Second, for the few respondents that preferred higher PES rates in the dry season, rather than in the wet season, it could potentially be because PES helped them to cope with the loss of livestock income during the 2008/09 drought.

Although this study's findings show that PES is a valuable supplement to livestock incomes of participating pastoral families, there are however, a few caveats. First, the reported income is a snapshot for a single period in 2008 and 2009. It would have been ideal to assess income for multiple years, but data was not available. Second, in pastoral communities, cash income (which is a flow as opposed to a stock in economic terms) often constitutes a relatively small component of a household's sum of economic assets and wealth, which are mostly held in livestock holdings and land assets (Galaty, 1981).

Finally, the land-use restrictions that exclude settlements and limit livestock grazing in the OOC mean that providers are faced with a trade-off: on the one hand, they derive high financial benefits by allocating their land to the conservancy; on the other, the PES conditions limit their herding of livestock, a core traditional pastoral practice with the potential to undermine their adaptive capacity.

In terms of human capital, access to technology and infrastructure, PES has both direct and indirect effects. The direct effects are evident among providers in the WLP who benefited from training on spatial data collection and participatory mapping, and thus gained valuable skills and knowledge (TechNews Africa, 2007). In the OOC, the direct effect of technology was evident through the mechanization of fodder harvesting and storage, and the construction of water tanks for the local community. The indirect effects are evident in the use of PES income to build human capital. In the WLP, for example, the highest proportion of expenditure from PES-generated income was allocated to education. This could be because the scheme is primarily designed to promote the education of children (Gichohi, 2003, Nkedianye, 2004).

Although the PES recipients have been involved in decision making, in terms of empowerment and local governance, there are also challenges that come with PES implementation such as the disempowerment of the landless, restriction to access of women to collect water and firewood and intra-household income distribution issues. Nevertheless, the fact that PES has opened up new avenues for empowerment of pastoral communities is particularly significant, given that these communities have previously been excluded from decision-making processes affecting their land use (Lenaola, Jenner and Wichert, 1996).

The effects of PES on institutions relevant to climate change adaptation

The literature on PES and climate change adaptation suggests that PES can lead to the creation of new institutions, or change existing ones in ways that have an impact on wider society, including its adaptive capacity (Wertz-Kanounnikoff et al., 2011). Rural institutions and organizations, in particular, shape the effects of climate hazards on livelihoods in three important ways: they structure environmental risks and variability (and thereby the nature of climate impacts and vulnerability); they create the incentive frameworks through which individual and collective actions unfold; and they act as channels through which external interventions reinforce or undermine existing adaptation practices (Agrawal, 2008).

In both the WLP and the OOC, pastoral providers and the ES users formed local institutions to promote collective action and to ensure effective participation in both the PES schemes and in wider policy and governance issues. More specifically, these institutions are affecting one or a combination of the five basic coping strategies and adaptation responses in the context of environmental risks to livelihoods: mobility; storage; diversification; communal pooling; and market exchange.

The PES-promoted land-use practices can either facilitate or hinder mobility which is a key adaptation strategy for pastoralists in the African rangelands (Niamir-Fuller, 1999). In the case of the WLP, for example, where pastoralists are paid to avoid crop production in marginal lands and to avoid fencing their plots, the open rangelands for wildlife realised through PES also facilitates the movement of pastoralists and their livestock in search of water and pasture during dry seasons, and particularly in drought periods. In the OOC, however, the periodic restrictions and limited access to livestock grazing inside the Conservancy may constrain livestock mobility and undermine the adaptive capacity of pastoralists (Birner and Osano, 2012).

In the OOC PES scheme, projects for storage of water and fodder have been introduced. The off-site fodder storage emerged as a response to the acute shortage of pasture during the 2008/09 drought that led to intense pressure on the OOC to allow unrestricted livestock grazing inside the conservancy land. With the funding support from the OOC Trust, the *Ol Purkel Ltd*. introduced the hay-baling project to harvest and store fodder to be used by providers or sold commercially in periods of drought in the future (Personal communication, Rob O'Meara, General Manager, *Ol Purkel Ltd*., June 17, 2011). This intervention is in line with the Kenya National Climate Change Response Strategy, which recommends the harvesting and storage of fodder by pastoral herders for use during dry seasons as a climate change adaptation strategy in the rangelands (Government of Kenya, 2010).

Local institutions are also influencing the way communal resource pooling is conducted. In the WLP, for example, landowners' membership based local organisation, the *Kitengela I'lParakuo Land Owners Association* (KILA) provides the vehicle through which both providers and non-providers alike can lobby to influence land-use planning, regulation and management. In the OOC, a land committee was established to represent and negotiate with the users' representatives on diverse issues, including payment rates, contracts, and conservancy regulations, and to act as an intermediary by signing land lease agreements with individual landowners and a collective agreement with the users. In this case, these institutions are influencing the way landowners adapt and respond to changes in land use through communal pooling and joint action.

In both PES schemes, institutional linkages — both vertical and horizontal — have been established to channel external support to the communities and to promote multi-level, inter- and cross-sectoral linkage participation in ways that strengthen the adaptive capacity of providers. Both inter-sectoral and cross-scale linkages are highly relevant for the adaptation and resilience of socio-ecological systems; the former to ensure policy coherence across sectors, and the latter to ensure coherence in decision making across scales (Robinson and Berkes, 2011, Wertz-Kanounnikoff et al., 2011). Overall, recognizing the scale factor in these PES projects can help to strengthen the resilience of the socio-ecological systems in which pastoral landowners operate, leading to increased collaboration, networking and connection with institutions that can provide land management and resource support in an increasingly globalized world (Galvin, 2009: 192) and generate solutions that are more politically and ecologically sustainable (Cash and Moser, 2000).

Both of these PES schemes face several challenges moving forward. First, trade-offs involving PES income, pastoral livelihoods and other diversification options are inevitable in these emerging PES schemes. These trade-offs are addressed at a broader level in Chapter 3. Locally, land use restrictions to livestock grazing and settlements such the case of OOC generates “leakages” when PES participants settle on communal lands and/or illegally graze their livestock inside the Maasai Mara National Reserve. These leakages may negatively affect non-participants that rely on communal lands, and may undermine wildlife conservation inside the protected area in the Maasai Mara National Reserve. It is recommended that counteracting mechanisms for example fodder provision is undertaken for PES providers where access to grazing land is restricted through PES measures that are likely to increase participants vulnerability to climate variability and change(van de Sand, 2012).

Second, the PES schemes assessed exclude the landless because only those who hold title to land can enroll. It is therefore necessary to understand in detail the inequities associated with PES implementation. These are addressed in Chapter 5 for the WLP in Athi-Kaputie Plains and Chapter 6 for the OOC PES scheme in the Maasai Mara. Finally, the question of the financial sustainability of these schemes remains a concern. Government and public funding from NGOs such as those that support the WLP are not indefinite and funding for “user-financed” schemes such as the OOC model relies on the tourism industry, which in Kenya is highly susceptible to financial and political shocks .This issue is addressed in the Conclusion chapter.

Conclusion

This chapter assessed the role of PES in adaptation to climate change and poverty among pastoral communities based on two schemes in southern Kenya Maasailand ASAL areas. First, it proposed a conceptual framework for the linkages between PES and EBA to climate change in Kenyan rangelands. Thereafter, it assessed the severity and frequency of droughts and considered the implications of drought and the role of PES on pastoral adaptive capacity, assessed the effects of PES on the adaptation of pastoral environmental service providers and on the local institutions relevant to adaptation.

In the conceptual framework of the linkages between PES and EBA to climate change, this study identified three potential pathways: changes in land tenure policies from communal to private ownership that drive transitions from open to closed rangeland states; land-use changes that constrain the mobility of wildlife and livestock across the landscape and increase the vulnerability of pastoral system as a result of low adaptive capacity; or PES for wildlife conservation and tourism, that may modify pastoral land management by promoting land-use practices that support open rangelands, to enhance the adaptive capacity of pastoral landholders.

Droughts in the two sites have increased in severity and frequency. The income from PES was found to be critical during droughts because it buffers pastoral families from fluctuating livestock income thereby helping them overcome short-term liquidity constraints arising from the high livestock mortality and low market prices. While PES can serve as a drought coping and risk mitigation mechanism in the short term, it remains unclear what role PES can play in relation to climate change in the long term.

The PES effects of ES providers' adaptation were also identified. The first is economic assets and wealth, where PES is seen as a relatively stable and predictable income source for pastoral households in the short-term within a context of high fluctuations in livestock income; PES may provide a safety net to buffer households when livestock income declines during droughts. Second, PES influences human capital, access to technology and infrastructure. Human capital can be enhanced directly through training and educational programs, or indirectly when PES income allows pastoralists to invest in education and health. Likewise, PES implementation can also lead to investments that increase access to technology and infrastructure to improve basic infrastructure, such as water and food storage.

Third, PES can alter ways in which local community are empowered empowerment and influence their participation in local governance process. While it can support pastoral landowners to participate in decision making, and create opportunities for positive adjustments in the governance and management of land use to the benefit of the landowners, it also can lead to the disempowerment of sections of local communities such as women by denying them access to critical natural resources.

Finally, PES was found to have three main effects on institutions relevant to climate change adaptation. First, apart from establishing new rules and regulations that influence patterns of pastoral land use and management, the implementation of PES has also led to the creation and strengthening of local collective action institutions that shape how local adaptation responses and coping strategies in terms of mobility, storage, diversification, communal pooling and market exchange happen in locally and how communities respond to changes in climatic variability and land use restrictions.

Second, PES has created inter-sectoral linkages involving land, water, tourism, wildlife, environment, governance and other sectors. Third, it has also created cross-scale linkages, since the institutions involved in PES operate at different scales of governance, including local, national and international levels. There is however some concerns with regard to the equity implications, leakages and the financial sustainability of the PES schemes analysed. Research on these issues is recommended to suggest ways of harnessing the potential of PES in climate change adaptation in Kenya and other pastoral rangelands globally.

BRIDGE BETWEEN CHAPTERS 4 AND 5

Chapter 4 developed a conceptual framework for understanding the linkages between PES and ecosystem based adaptation in the Arid and semi-arid lands. It then analyzed the trends in drought occurrence and severity at the two sites over the last several decades, and reviewed the observed effects of PES on the adaptive capacity of pastoral environmental service providers and on the local institutions relevant to adaptation. The chapter was based on a comparative analysis of the two PES programs operational in Maasai Mara Ecosystem (MME) and the Athi-Kaputiei Plains (AKP). The chapter found that drought events have increased in severity and frequency at the two study sites and that the drought in 2008-2009 in particular resulted in high livestock losses, and reduced cash income from livestock among the families surveyed. Chapter 4 concluded that in the short-term PES can serve as a drought coping and risk mitigation mechanism because it helps buffer participating pastoral families from fluctuating livestock income thus preventing these families from spiralling into poverty. The chapter concluded by discussing and highlighting the different roles of PES in climate change adaptation including both the positive and negative implications.

Chapter 5 and 6 focuses on the two PES schemes reviewed in this thesis in detail to generate empirical evidence regarding the actual effects of a publicly funded PES scheme (Chapter 5) and a user (market)-funded PES scheme (Chapter 6) on household poverty and inequality among pastoral. Chapter 5 is based on data from the AKP and uses the case study of the Wildlife Lease Program (WLP), a publicly funded PES scheme whereby pastoral land owners living on the wildlife dispersal area to the south of Nairobi National Park are paid US$ 10/ha/year to refrain from cultivation, land sales and sub-division, and to allow wildlife on their private land. In Chapter 5, I conduct an analysis of the PES institutional arrangements, assess the determinants of participation and intensity of participation in the WLP, and present the findings of the WLP's effects on poverty and inequality among both participating and non-participating households.

CHAPTER 5: POVERTY, INEQUALITY AND PARTICIPATION OF PASTORALISTS IN A PAYMENT FOR ECOSYSTEM SERVICE SCHEME ADJACENT TO A SEMI-ARID PROTECTED AREA IN SOUTHERN KENYA

Philip M. Osano, Mohammed Y. Said, Shem Kifugo, Jan de Leeuw, Nicholas Ndiwa, Hippolytte Affognon, Joseph Ogutu

Abstract

Payments for Ecosystem Services are being adopted in the Arid and Semi-Arid Lands (ASAL) across East Africa to promote ecotourism through wildlife conservation on private lands, and to reduce poverty among rural pastoral land users. We conducted an assessment of the Wildlife Lease Program (WLP), a PES scheme whereby pastoral land owners living on the wildlife dispersal area to the south of Nairobi National Park are paid US$ 10/ha/year to refrain from cultivation, land sales and sub-division, and to allow wildlife on their private land to determine its impact on poverty and inequality. Since 2000, the WLP provided payments totaling US$ 605,170 to 388 landowners that enrolled a total of 16,774 ha of land in the WLP PES scheme. Our analysis shows that close to three-quarters of all our sampled households are income-poor and nearly half are land-poor. Without accounting for the opportunity costs, the magnitude of the cash transfer is, on average, sufficient to close the poverty gap. Econometric analysis of our survey data shows that farm size and human capital significantly positively influence participation and intensity of participation, with the former also significantly influenced by asset endowments, wildlife depredation, and pasture conditions. These findings suggest that, in terms of participation, the WLP is not strictly pro-poor. First, except for poor households occupying areas with low quality pasture, households with a higher probability of participating in the scheme are generally those that own larger farms (land size) and have higher asset endowments, all of which are associated with relatively low levels of poverty. Second, poor households with low incomes and small land assets face greater constrains in the WLP because they have lower participation intensity. Despite these reservations in terms of poverty, the WLP was found to be the most equitable of all income sources for participating households and is also an invaluable source of income diversification during droughts when livestock income may decline.

Introduction

Biodiversity contributes to human well-being through supporting, provisioning, regulating, and cultural ecosystem services (Millennium Ecosystem Assessment, 2005a), but biodiversity loss and deterioration of ecosystems remain major concerns (Secretariat of the Convention on Biological Diversity, 2010a), that has recently been heightened by the failure to achieve the global target of significantly reducing the rate of biodiversity loss by 2010 (Butchart et al., 2010). Biodiversity and habitat loss are severe and continue unabated in the low-income, biodiversity-rich tropical countries (Brooks et al., 2002), which have low capacity for biodiversity management (Barrett et al., 2001).

The high poverty levels and dependence of rural poor on ecosystem services in tropical countries (UNEP-WCMC, 2007, Turner et al., 2012) justify coupling biodiversity conservation with poverty reduction to achieve win-win outcomes (Sachs et al., 2009, Timmer and Juma, 2005), a difficult feat in practice; so the challenge remains (Agrawal and Redford, 2006, Adams et al., 2004, Roe et al., 2010). This challenge is greatest in Africa where high poverty levels abound alongside a rich biodiversity heritage, including spectacular wildlife species diversity (Collier, 2007, UNEP, 2006).

Shrinking populations and contracting geographic ranges of terrestrial wildlife species are of primary conservation concern in Africa (Loh and Wackernagel, 2004), particularly for large mammals whose populations within and outside protected areas have declined drastically since the 1970s (Caro and Scholte, 2007). For example, 69 species of large mammals inhabiting 78 parks declined by 59% during 1970-2005, in West and mostly in East Africa (Craigie et al., 2010, Scholte, 2011), where dramatic declines have recently been documented in Kenya (Western et al., 2009, Ogutu et al., 2011), and Tanzania (Caro, 2008).

A leading cause of biodiversity loss, including declines in the populations of large mammals, is habitat alteration (Millennium Ecosystem Assessment, 2005a), through land use changes (Kiss, 2004b). In East Africa's savannas where wildlife and pastoralism have co-existed for centuries, wildlife is progressively displaced by agriculture and other non-conservation compatible land uses (Homewood et al., 2001, Norton-Griffiths and Said, 2010).

An early strategy to protect large mammals was the establishment of protected areas (PA's) which now exceed 1,050 and cover approximately 7% of Africa's landmass (UNEP, 2006). PA's however, face ecological, economic and political challenges that constrain their effectiveness (Bruner et al., 2001, Brandon et al., 1998, Alers et al., 2007). Ecologically, most PA's are too small, were not designed based on biodiversity assessments nor threats to habitats, and consequently, do not meet the full habitat requirements of many large herbivores, which disperse seasonally to adjacent lands (Newmark, 2008, Fynn and Bonyongo, 2010). Economically, PA's are underfunded with limited capacity for monitoring and law enforcement (Wilkie et al., 2001, Bruner et al., 2004) and foreclose future land use options, generating significant opportunity costs (Norton-Griffiths and Southey, 1995). Politically, the creation of PA's caused the eviction of former occupiers or right holders in land (Schmidt-Soltau and Brockington, 2007), mostly exclude local participation (Adams and Hutton, 2007, Hulme and Murphree, 1999), and rarely contribute sufficiently towards the welfare needs of neighboring communities (Naughton-Treves et al., 2005, Redford and Fearn, 2007, Galvin and Haller, 2008).

These limitations in the protected area approach led to a paradigm shift towards decentralization of conservation (Berkes, 2004), which recognizes the role of and seek benefits to local communities in wildlife conservation. Decentralized interventions, such as Community-Based Conservation (Western and Wright, 1994), Community-Based Wildlife Management (Child, 1996), Integrated Conservation and Development Projects (Wells and McShane, 2004, Wells et al., 1992) and Community Based Eco-tourism (Honey, 2009, Kiss, 2004a), focus strongly on rural livelihoods through provision of alternative products, incomes, and social benefits (Kiss, 1990) but are also rarely effective in advancing conservation or delivering lasting local development benefits (Hackel, 1999, Kellert et al., 2000, Newmark and Hough, 2000).

More recently, biodiversity markets (Jenkins et al., 2004, OECD, 2003) based on direct payments (Ferraro and Kiss, 2002, Milne and Niesten, 2009) are increasingly applied to wildlife management on private lands to promote biodiversity-friendly land use practices (Jack et al., 2008, Kiss, 2004b). These market mechanisms are a form of Payments for Ecosystem Services (PES), defined as "(1) a voluntary transaction in which (2) a well defined environmental service (or land use promoting that service) (3) is "bought" by a buyer (4) from a provider (5) if and only if the provider continuously secures the provision of the service"(Wunder, 2005). PES schemes are predicated on the principle that ecosystems provide services that support human well-being (Millennium Ecosystem Assessment, 2005b) and that beneficiaries of these services should pay those (e.g. rural farmers) whose land use activities generate ecosystem services (Wunder, 2005, FAO, 2007).

Payments for Ecosystem Services are expanding rapidly in developing countries partly because of their appeal to generate new finances for conservation (Wunder and Wertz-Kanounnikoff, 2009, Wunder et al., 2008), and potential for poverty reduction (Lipper et al., 2009, Ravnborg et al., 2007, Grieg-Gran et al., 2005, Pagiola et al., 2005). Biodiversity PES schemes support the purchase of high value habitats, payments for access to species and habitats, conservation management and biodiversity-conserving businesses (Jenkins et al 2004). In wildlife conservation, PES is presented as a suitable policy tool for limiting habitat destruction by controlling access to land critical for conservation (Pagiola, 2003), and for mitigating human-wildlife conflicts, particularly livestock predation by large carnivores (Dickman et al., 2011, Nelson, 2009).

The use of PES to halt habitat loss and degradation that lead to decline in wildlife populations is becoming common across East and southern Africa, including Tanzania (Nelson et al., 2010, Sachedina and Nelson, 2012), Zimbabwe (Frost and Bond, 2008) and Namibia (Naidoo et al., 2011). In Kenya, wildlife PES mostly involve conservation land leases in private individual, communal or public trust lands, particularly adjacent to protected areas in the arid and semi arid lands (ASAL), which cover 88% of Kenya's landmass and contain almost 90% of Kenya's wildlife (Norton-Griffiths and Said, 2010).

The Arid and Semi-Arid Lands (ASALs) are mainly inhabited by pastoral communities, majority of whom live in remote and marginalized areas under challenging economic conditions following decades of economic marginalization and political exclusion engendered by policies that undermined, rather than supported pastoralism (Galaty, 1999, Schneider, 1990, Fratkin, 1997). Chronic poverty is prevalent among pastoral communities characterized by perennial food insecurity (dependence on relief food), low financial incomes and minimal access to basic social services such as education and health (Okwi et al., 2007, Little et al., 2008, Anderson and Broch-Due, 1999). While pastoral households depend primarily on their livestock, many are diversifying into non-livestock income-generating activities as part of a multi stranded livelihoods (Homewood et al., 2009b).

Changes in land tenure have led to land privatization and sub-division (Galaty, 1994a) creating new opportunities and challenges in pastoral land use and management (Behnke, 1994). Thus wildlife PES is seen as an opportunity for income diversification (Homewood et al., 2009b).

Although little is currently known of the implications of PES on poverty among pastoral households, experience in other socio-ecological settings shows that PES generates both positive and negative impacts affecting participants, non-participants and ecosystem service users differently (Grieg-Gran et al., 2005, Wunder, 2008, Zilberman et al., 2008, Leimona and Joshi, 2009). A recent study that modeled different scenarios for introducing PES to pastoral landowners adjacent to the Amboseli National Reserve in Kenya concluded that wildlife PES can concurrently protect elephant habitat from cropping while also reducing poverty among participating households (Bulte et al., 2008a). However, this kind of win-win scenario has not been empirically demonstrated.

It is thus germane to assess existing wildlife PES schemes among pastoral land users to determine their poverty effects.

This paper contributes to filling the current knowledge gap regarding PES and poverty by assessing the Wildlife Lease Program (WLP), a PES scheme on private pastoral lands adjoining the Nairobi National Park in southern Kenya. We address four questions:

1) What is the nature of the design and implementation of the WLP?
2) What is the level of poverty and inequality among the households in AKP?
3) What are the determinants of participation, and intensity of participation in the WLP?
4) What are the effects of the PES on household poverty, inequality, income and expenditure?

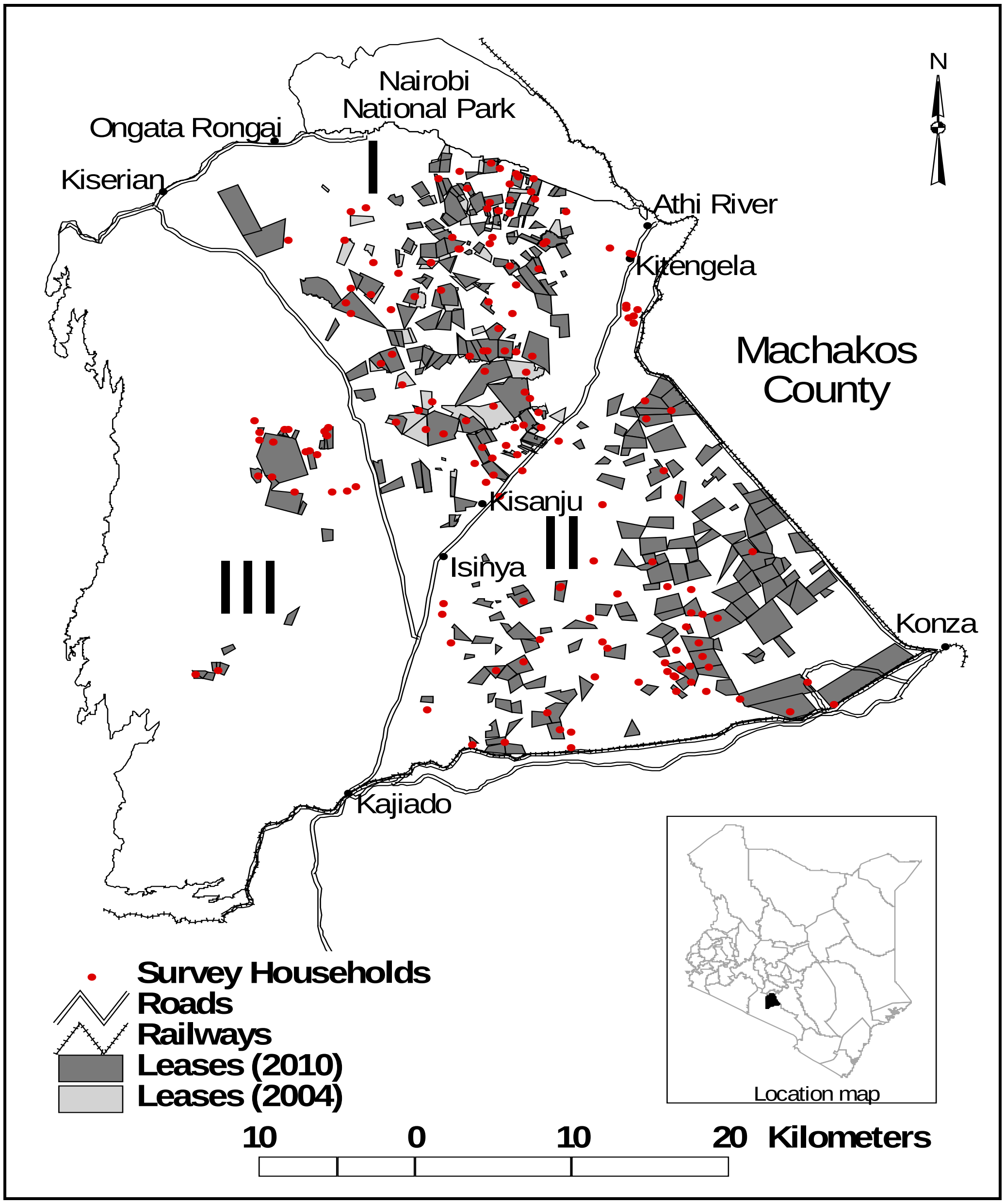
The remainder of the paper is organized as follows. Section 2 presents the methods and a description of the WLP. Sections 3, 4 and 5 present the results, discussion and conclusions, respectively.

Methods

Study area

The study area is in the Athi-Kaputiei Plains (AKP) in South-western Kenya, which is a critical wildlife dispersal area for the Nairobi National Park (NNP: 117 km^2) situated on the southern periphery of Nairobi metropolis (Figure 5.1) and is part of the larger Athi-Kaputiei Ecosystem (2000 km^2, (KWS, 2004). The area is dry and the annual rainfall increases from 500 mm in the southeast to 800 mm in the northwest. The vegetation is predominantly wooded *Acacia/Balanites/Themeda* grassland, with gallery forests of *A. xanthophloea* and small forest patches of *Croton macrostachys* and *Olea Africana* along rivers (Reid et al., 2008).

Figure 5.1 Map of the Athi-Kaputiei Plains showing the location of the surveyed households, the land parcels enrolled in the Wildlife Lease Program (WLP) in 2004 and 2010, Nairobi National Park (NNP) and Triangles I, II and III. Nairobi Metropolis is on the northern edge of NNP.



The NNP and the wider ecosystem support 24 species of resident large mammals, with wildebeest (*Connochaetes taurinus* Burchell) and zebra (*Equus Burchelli* Gray) constituting about half of all the wildlife. The park itself is too small to support viable populations of most of the large herbivores. It is fenced on all sides except on the southern boundary which is open to enable wildlife especially wildebeest, zebra and eland (*Taurotragus oryx*) that use the park as a dry season (June-November) refuge, to disperse to the AKP in the wet season (March-May) for forage and calving (KWS, 2004). While outside the park, the wildlife use private land owned mostly by Maasai pastoralists, who bear costs arising from competition with their livestock for forage and water, injury and killing of humans and livestock, transmission of infectious diseases to livestock, and damage to their crops, but receive little or no compensation from the wildlife authorities. This has led to human-wildlife conflicts and hostility between the park authorities and the local community (Khisa, 2001).

The NNP and AKP ecosystem face two major conservation challenges. First, the ecosystem is severely degraded (Gichohi, 1996), adversely affecting the park, adjacent private lands, and the viability of wildlife populations. The ecosystem is listed in the Draft Wildlife Bill (2009) as “Critically Endangered” owing to the severe degradation of its structures and functions and the imminent threat of its irreversible transformation (Republic of Kenya, 2009). Second, wildlife populations inside the NNP and within the AKP have declined drastically in the last 30 years. In AKP, the overall population declined by 72% during 1972-2002. Of the two most common herbivore species that migrate between the park and the adjacent lands, wildebeest declined by more than 90% but zebra did not decline during 1972-2002 (Reid et al., 2008). The numbers of cattle, sheep and goats fluctuated widely, declining sharply during the droughts of 1994-1996, 1999-2000, and 2009-2010 but showed no overall trend (Ogutu et al., Submitted).

The extreme declines in wildlife populations are attributed to habitat loss, fragmentation and degradation caused by land use changes, including conversion of grasslands to croplands, establishment of permanent settlements, and urban expansion, land sub-division, fencing and human population growth (Reid et al., 2008). Land in AKP was privatized in mid-1970s following the creation of the group ranches whereby property rights to land was held collectively, but in 1988, group ranch land-subdivision to plots with individual property rights to land started in AKP (Rutten, 1992).

Data

The data were obtained from primary and secondary sources. The primary sources were (1) a survey of 166 households from November 2009 to January 2010 selected by random sampling stratified program enrolment (Participating and non-Participating households). The sample comprised 86 participants in the WLP and 80 non-participants. The survey elicited detailed information on household socio-economic and demographic characteristics, land use and involvement in the WLP. We recorded Global Positioning System (GPS) points for all the surveyed households and used these to generate the spatial variables used in our analysis. (2) Semi-structured and informal interviews with landowners, WLP managers, funders, community leaders and key informants.

The secondary data used included the databases provided by The Wildlife Foundation (TWF), which administers the WLP, and International Livestock Research Institute (ILRI). The TWF database includes 10-year (2000-2010) data on the WLP, including enrolment, funding and payment records, monitoring and evaluation reports, and land lease contracts. The ILRI database contains geo-spatial data on land cover and land use, land parcels, fence maps (for 2004 and 2010), water sources (rivers, dams), urban areas, social amenities (health centers, schools, markets), infrastructure (roads, railway) and spatial poverty and inequality data (for 1999) from the Kenya government (Central Bureau of Statistics, 2003, Central Bureau of Statistics, 2005).

The Wildlife Lease Program

The WLP was started in April 2000 and it targets an area of 60,000 acres (24,291 ha) in the AKP. The WLP has two objectives; to keep the seasonal wildlife dispersal areas and migration corridors open to ensure the viability of the NNP and its biodiversity by enabling continued wildlife movements between the NNP and the AKP; and to enhance the economic security and quality of life of the participating households (Gichohi, 2003, World Bank, 2008).

Households participating in the WLP voluntarily enroll their land and are paid an annual fixed fee of KES 300 per acre (US$ 10 per hectare in 2002) directly to their bank accounts and the cheques are disbursed at a public ceremony (Gichohi, 2003). The WLP requires the enrolled land user to manage land for wildlife and livestock grazing and prohibits fencing, quarrying, cultivation, sale or sub-division. To be eligible for the WLP, a land owner in AKP must have open and unfenced land (fences must be dismantled) and present a title deed is as proof of ownership with a copy is retained by TWF for recording and monitoring.

The WLP payment rate reflect the estimated economic returns to existing legal land use options in AKP by 2002 (Kristjanson et al., 2002) with wildlife income restricted to non-consumptive uses legally allowed at the time (Norton-Griffiths, 1996). A contingent valuation study conducted prior to the establishment of the WLP in 1999 to assess how much money landowners in Kitengela which comprises the northern portion of AKP adjoining the Nairobi National Park were willing to accept as compensation for wildlife losses incurred in their land was deemed questionable as landowners asked for an average compensation of US$ 920 acre per year (US$ 2,260 per hectare per year) (Mwangi and Warinda, 1999) which was thought to be grossly inflated.

Data analysis

The survey data were analyzed using MS-Excel, SPSS and SAS softwares, and geo-spatial analysis conducted in ArcGIS 9.3 (ESRI, Redlands, California). We converted the WLP payments to US dollar equivalents, and standardized to the year 2005 using the Kenya GDP Deflator obtained from the World Bank.[1] We calculated descriptive statistics for the survey data to compare participating and non-participating households. We excluded from analysis six households that dropped from the WLP and two whose questionnaires were incomplete, resulting in 158 households.

[1] World Bank website online http://data.worldbank.org/indicator/NY.GDP.DEFL.ZS

The WLP scheme design and implementation

We evaluated the WLP based on four elements of an 'idealized' PES scheme: voluntary transaction; actors; ecosystem services; and conditionality and monitoring (Wunder, 2005, Wunder, 2007).

Poverty and inequality assessment

We mapped the distribution of surveyed households in relation to poverty (rate, density and gap) using 1999 poverty data (Central Bureau of Statistics, 2003), and then assessed household poverty and inequality based on cash income and land asset as indicators. The *poverty rate* (head count index) measures the percentage of people falling below the poverty line, and reflects how widespread poverty is within a given area; *Poverty gap* measures the depth of poverty, and is calculated by adding up all the shortfalls of the poor (ignoring the non-poor) and dividing by the total population; *Poverty density* is defined as the number of poor people living in a given geographic area (e.g. per km^2) (Coudouel et al., 2002, Central Bureau of Statistics, 2003).

We calculated income poverty by dividing the gross household income (2009) by the household size expressed in terms of adult equivalent (AE), to obtain the annual per capita gross income, which we converted to monthly income in tandem with the official Kenyan poverty line. The concept of adult equivalent (AE) is based on differences in human nutrition requirements according to age, where; <4, 5-14 an > 15 years of age are equivalent to 0.24, 0.65 and 1 AE respectively (Kristjanson et al., 2002). We then classified all households with an income equal to, or below, the monthly Kenyan rural poverty line of KES 1,562 per capita as poor, and the remaining as non-poor. The Kenya poverty line (KES 1,562 in rural areas and KES 2,913 in urban areas per person per month) is based on estimated expenditures on minimum provisions of food and non-food items.[2]

[2] Details are available online: http://opendata.go.ke/Poverty/District-Poverty-Data-KIHBS-2005-6/pnvr-waq2

Land based PES schemes provide payments based on the land area enrolled, making it critical to assess their poverty implication in terms of land ownership and distribution (Pagiola et al., 2005) as the landless may be excluded, and the 'land poor' households may face participation constraints (Grieg-Gran et al., 2005). We established land poverty by calculating the mean, mode and median values for land ownership among surveyed households, and considered those with land equal to or less than the median value of 80.94 ha as land-poor.

To assess inequality, we first mapped the distribution of surveyed households in relation to inequality (gini-coefficient) using per capita expenditure data for 1999 (Central Bureau of Statistics, 2005). Next, we then computed the gini-index for cash income (2009), livestock (cattle, goats and sheep) and land (Araar and Duclos, 2009) and then calculated the differences between the gini-coefficient of participants and non-participants based on standard error estimates (Russell, 2009). Lastly, we calculated and then compared the percentage coefficient of variation (CV) for the different sources of household income. High %CV scores indicate high levels of inequality and vice-versa.

Dependent and explanatory variables for regression analysis

The predictor variables (Table 5.1) were selected from factors hypothesized to influence landowner participation and intensity of participation in PES based on the literature and discussions with landowners and local community. These variables describe the characteristics of the households, the farming systems, and nature of information diffusion, and include farm size, income, credit, family size and labor availability, farmer age and education, accessibility of information, and have all been found relevant in previous empirical studies (Pagiola et al., 2005, Pagiola et al., 2010, Pagiola et al., 2008, Zilberman et al., 2008, Uchida et al., 2007, Gauvin et al., 2009, Mullan and Kontoleon, 2009).

Table 5.1 Descriptive statistics for variables used in the logistic regression analysis (N=158).
Source; Lead authors' survey

Variable	Description	Percentage (%)	
Categorical variables		**1**	**0**
Dependent variable			
WLP status	1=participating; 0=non-participating	54.43	45.57
Independent variables			
Gender	1=male; 0=female	72.78	27.22
Marital status	1=married; 0=not married	84.54	16.46
Education	1=some education; 0=no education	51.90	48.10
Employment	1=employed; 0=unemployed	51.27	48.73
Access to credit	1 = access to credit; 0= lacks access to credit	43.04	56.96
Wildlife predation	1=yes; 0=no (3 year period from 2007-2009)	66.46	33.54
Poverty Income (09)[a]	1=non-poor ; 0=poor	33	67
Poverty_land[a]	1=non-poor (land >80.9ha); 0=poor (land<=80.9ha)	57	43
Continuous variables		**Mean (SD)**	**Median**
Dependent variable			
Proportion of land allocated to PES[1]	Area enrolled in PES as a proportion of total land area (%)	38.19 (43.44)	24.93
Independent variables			
Total adult labor	Number of all adults above 18 years in the household	4.70 (2.99)	4.00
CAI	Composite asset index (CAI)	2.75 (0.70)	2.76
Child dependency ratio	Ratio of number of persons younger than 18 years to the total adult labor in the household	0.98 (0.95)	0.71
Cash Income	Gross cash income in 2009 (in KES)	84,781.71 (98,926.08)	57,800.00
Current plot area	Current area of plot of land (ha)	119.98 (141.36)	80.94
Distance to town	Distance to the nearest major town (Km)	7.28 (3.40)	6.98
Distance to road	Distance to the nearest all-weather road (Km)	2.18 (1.88)	1.55
Distance to school	Distance to the nearest primary school (Km)	4.2 (2.06)	4.11
NDVI_mav4	NDVI moving average over 72 months (2004 - 2009)	0.23 (0.07)	0.20

[a] This applies to households participating in the Wildlife Lease Program (WLP)

We also included variables characterizing the decisions of the WLP administrators regarding who should participate since in PES schemes which are oversubscribed, the final choice of whether to enroll a particular land owner is made by the PES scheme administrators (Mullan and Kontoleon, 2009). Previous literature on the WLP shows that it is oversubscribed (Gichohi, 2003, Sikand, 2007, Nkedianye, 2004), a factor further confirmed from our surveys which found 82% of the current non-participants willing to join the WLP.

Based on interviews with the TWF and funders, and a review of literature on the WLP (Kristjanson et al., 2002, Nkedianye et al., 2009, Nkedianye, 2004, Gichohi, 2003, Rodriguez et al., 2011a, Reid et al., 2008), we identified three observable factors expected to determine the probability that TWF would enroll a particular plot of land in the WLP. First, the selection criteria relating to wildlife conservation include proximity to the park. As households with land closer to the park are more likely to experience higher levels of livestock depredation by large carnivores, we thus included wildlife predation as an explanatory variable.

Second, a secondary objective of the WLP is poverty reduction (World Bank, 2008), so we computed a composite asset index (CAI) using principal component analysis (Abeyasekera, 2005, Vyas and Kumaranayake, 2006), and included both income and CAI as explanatory variables to test whether the WLP is being targeted at poor households identified on the basis of both income and assets. The CAI is based on six durable assets (cell phone, bicycle, television, radio, motorcycle, and vehicle), land and house quality (Table 5.2). In the case of house quality, we took a binary value of 1, if a household owned a particular asset or 0 if not, for all the six durable assets classified household quality as either ‘un-improved’ or ‘improved’ based on the materials used in construction. The houses constructed using purely traditional materials, or in combination with iron sheets, or iron sheets alone were classified in the ‘un-improved’, whereas all houses constructed using cement were classified as ‘improved’.

Table 5.2 Result of the principal component analysis (PCA) to develop a composite asset index for household wealth/poverty status.
Source; Lead author's survey

Rotated Component Matrix[a]	
Asset Categories	**Component**
Ownership of a vehicle	0.750
Ownership of a Television	0.746
Quality of the household	-0.654
Ownership of a cell phone	-0.050
Ownership of a bicycle	0.024
Ownership of a motorcycle	0.201
Ownership of a radio	0.140
Land above or below 80.94 hectares	-0.075

Extraction Method: Principal Component Analysis.

Rotation Method: Varimax with Kaiser Normalization.

[a]Rotation converged in 5 iterations.

The CAI was computed using the formula:

CAI=(0 0750 × vehicle) + (0.746 × TV) – (0.654 × housequal)–(0 050× cell phone) + (0.024 × bicycle) + (0.201× motorcycle) + (0.140× radio) – (0.075× land above_below80.94ha),

Furthermore, the poverty objective of the WLP is directed towards education and payments are disbursed during the start of school term to enable recipients to use the PES income on educational expenses such as school fees (Gichohi, 2003, Nkedianye, 2004). We therefore included distance to the nearest primary school as a proxy for access to education.

Third, for the TWF, monitoring is easier if enrolled parcels are easily accessible (located closer to roads, towns, etc), because some parts of the study area are remote with rough terrain making access difficult in the wet season. We therefore included distance to the nearest town as a predictor variable to account for ease of access.

Finally, due to the interest in the role of PES as a risk coping strategy during drought and taking into account the three major drought incidences recorded in the AKP in 1999-2000, 2005-2006, and 2009-2010, which led to huge livestock losses, with negative effects on pastoral livelihoods (Nkedianye et al., 2011, Osano, 2011), we hypothesized that drought effects may also influence household enrolment decisions and included the Normalized Difference Vegetation Index (NDVI) as an explanatory variable to proxy for drought conditions affecting grazing and pasture potential.

The NDVI is a remote-sensing based index ranging from 0 to 1 reflecting the greenness of vegetation with NDVI <0.20 – 0.25 for bare soil and dead vegetation and NDVI of 0.6 – 0.7 for green vegetation with canopy. It therefore indexes the amount of green vegetation produced in response to rainfall and reflects the lagged effects of rainfall and other influences in savannas. We obtained NDVI indices for the GPS points for each of the 158 households from the National Oceanic and Atmospheric Administration (NOAA) Advanced Very High Resolution Radiometer (AVHRR) images at a pixel resolution of 8×8 km^2 and then calculated a moving NDVI average over a 72-month window spanning 2004-2009 because 69% of all the WLP enrolments occurred between 2004 and 2010, and also to account for the 2005-2006 and 2008-2009 droughts.

Econometric model

We address two aspects of household participation. The first is whether a particular household participates in the WLP, and the second, restricted to WLP households only, relates to their intensity of participation. In the first case, we modeled participation as a binary outcome (Mullan and Kontoleon, 2009) using a logistic regression model to estimate the strength of determinants of the binary choice between participation (1) and non-participation (0) (Green, 2003).

In the second case, we modeled the intensity of participation by taking the percentage proportion of each household’s total land allocated to the WLP as the dependent variable. Here, our approach is similar to that of Pagiola and colleagues who also looked at the determinants of intensity of participation with a focus on whether poorer households are less able to participate than better-off households in PES schemes in Nicaragua (Pagiola et al., 2008) and Colombia (Pagiola et al., 2010) respectively. We run a generalized linear model (GLM) assuming a binomial error distribution and using a logit link function, and also included a random household effect in the model but dropped it since it lacked support based on the corrected Akaike's information criterion (AICC) (Burnham and Anderson, 2002). To account for potential nonlinearity, we included quadratic terms in the predictor variables, but again, these were not supported by the data hence were dropped from the model.

Financial income effects of the WLP on participating households

As long as participation is voluntary, there is a *prima-facie* assumption that PES will make recipients better off, otherwise, they will not participate (Wunder, 2008, Pagiola et al., 2005). A fundamental question for participants then is "Does participation make them better off?"(Wunder, 2008). The existing literature on income gains follows two approaches to answer this question. The first is to assess PES payments relative to the opportunity costs to participating households (Kosoy et al., 2007). The second is to assess PES payments as a proportion of household income or expenditure (Kosoy et al., 2007, Grieg-Gran et al., 2005). Estimating the opportunity cost is beyond the scope of this study, so we only assess the relative contribution of PES to the gross household income.

Results

Participating and non-participating households

Households participating in WLP were significantly different from non-participants in terms of size ($P< 0.01$), land owned ($P<0.05$), labor availability ($P<0.01$), wildlife predation ($P<0.05$), pasture conditions of household location (NDVI: $P<0.05$), and cash income ($P<0.1$) (Table 5.3). The average household size was 9.4 for participants and 7.0 for non-participants. The mean land holding was larger among participants (142 ha) than non-participants (90 ha), and participants earned, on average, KES 97,717 compared to non-participants that earned KES 70,215 in 2009 (Table 5.3).

Table 5.3 Summary statistics (mean) for the households surveyed in the Athi-Kaputiei Plains (standard deviations in parenthesis in columns (1) and (2)).
Source; Lead authors' survey.

	(1) Participants	**(2) Non-Participants**	**t-statistic (p-value)**
Continuous variables			
Household size (persons)	9.38 (6.34)	6.99(3.60)	2.749 (0.007)***
Land owned (ha)	141.59 (169.31)	90.31(93.69)	2.293 (0.023)**
Gross income in 2009 in KES (mean)	97,717.44 (86,422.06)	70,215.28 (111,106.35)	1.749 (0.082)*
Composite asset index (CAI)	2.819 (0.690)	2.677 (0.703)	-1.289 (0.199)
Household adult labor (persons)	5.337 (3.263)	3.944 (2.449)	-2.985 (0.003)***
Child dependency ratio (ratio)	0.943 (0.925)	1.034 (0.982)	0.5988 (0.550)
Cattle in 2009 (TLU)	25.78 (33.69)	21.15 (35.24)	0.843 (0.400)
Shoats in 2009 (TLU)	17.45 (15.49)	14.22 (14.21)	1.354 (0.178)
Distances (km) to nearest major town	7.03 (3.15)	7.60 (3.76)	-1.053 (0.294)
Distances (km) to nearest road	2.32 (2.04)	2.02 (1.64)	1.010 (0.314)
Distances (km) to nearest primary school	4.11 (2.09)	4.38 (2.06)	-0.801 (0.425)
NDVI_mav4	0.219 (0.066)	0.243 (0.071)	2.159 (0.032)**
Categorical variables			**Chi-square test (DF)**
Education level (%)	46	54	4.497 (1.00)
Employment (%)	52	50	0.771(1.00)
Access to credit (%)	41	46	0.516 (1.00)
Wildlife predation (%)	74	57	0.021** (1.00)
No. of household in sample (n)	86	72	

Notes: **P*<0.1; ***P*<0.05; ****P*<0.01.

Design and implementation of the WLP

The WLP started in 2000 with 18 households allocating 688 ha which increased to 357 households allocating 16,774 ha of land by September 2010 (Figure 5.2a; b). This represents 69% of the target area and 8% of the ecosystem excluding NNP. During this 10-year period, the WLP disbursed a total of US$ 605,170 (in US$ 2005 equivalent) directly to the participating households (Figure 5.2a).

The temporal growth in enrolment reflects two main phases; the first from 2000 to 2003 during which the number of enrolled households increased from 18 to 117, and the second, from 2007 to 2010 during which the enrolled households increased from 111 to 357 (Figure 5.2a). Notably, only 20% of the total land (4,000 ha) was enrolled in the first eight years from 2000 to 2008 and the remaining 80% (12,150 ha) in the 2 years from 2008 to 2010.

Figure 5.2 A. Household enrolment in the Wildlife Lease Program (WLP) PES scheme and the total annual PES payments to households (in 2005 'US$ '000) in the period 2000 to 2010; **B.** The total land area enrolled in the WLP PES in the period 2000 to 2010.
Data source; The Wildlife Foundation.

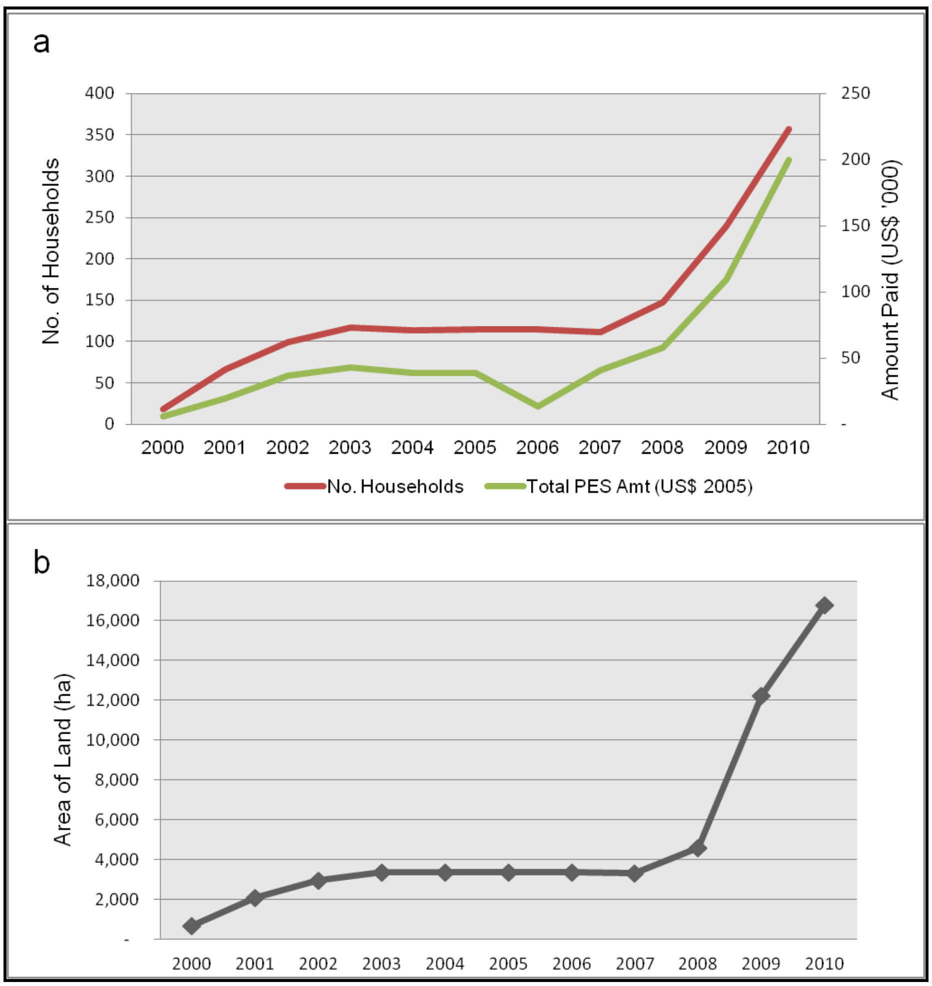


Table 5.4 shows respondents' perception of some aspects of the WLP payments. Although 67% of respondents indicate that they were provided with sufficient information prior to enrollment, all (100%) reported that they were not involved in setting the annual lease price of US$ 10 per hectare, the majority (94%) were not aware who set the lease price and very few (2%) knew how the price was determined. Currently, 98% of the participants consider the rate as insufficient.

Table 5.4 Perceptions and views of landowners regarding the PES payment features and contract arrangements in the Wildlife Lease Program (WLP) (n=86 landowners). *Source;* Lead author's survey.

Statement	**Yes (%)**	**No (%)**
Did you participate in setting the amount that is paid in the wildlife lease program (WLP) of KES 300/acre/year?	0	100
Do you know how the WLP lease price of KES 300/acre/year was set?	2	98
Do you know who set the WLP lease price of KES 300/acre/year?	6	94
Do you consider the current WLP lease of KES 300/acre/year to be sufficient?	2	98
Do you consider the information provided to you before joining the wildlife lease program (WLP) as sufficient?	67	19
Are you interested to stay in the wildlife lease program (WLP) in the next five years?	92	8
Would you accept a lease contract that is legally binding?	42	58

KES: Kenya Shillings

Voluntary transaction

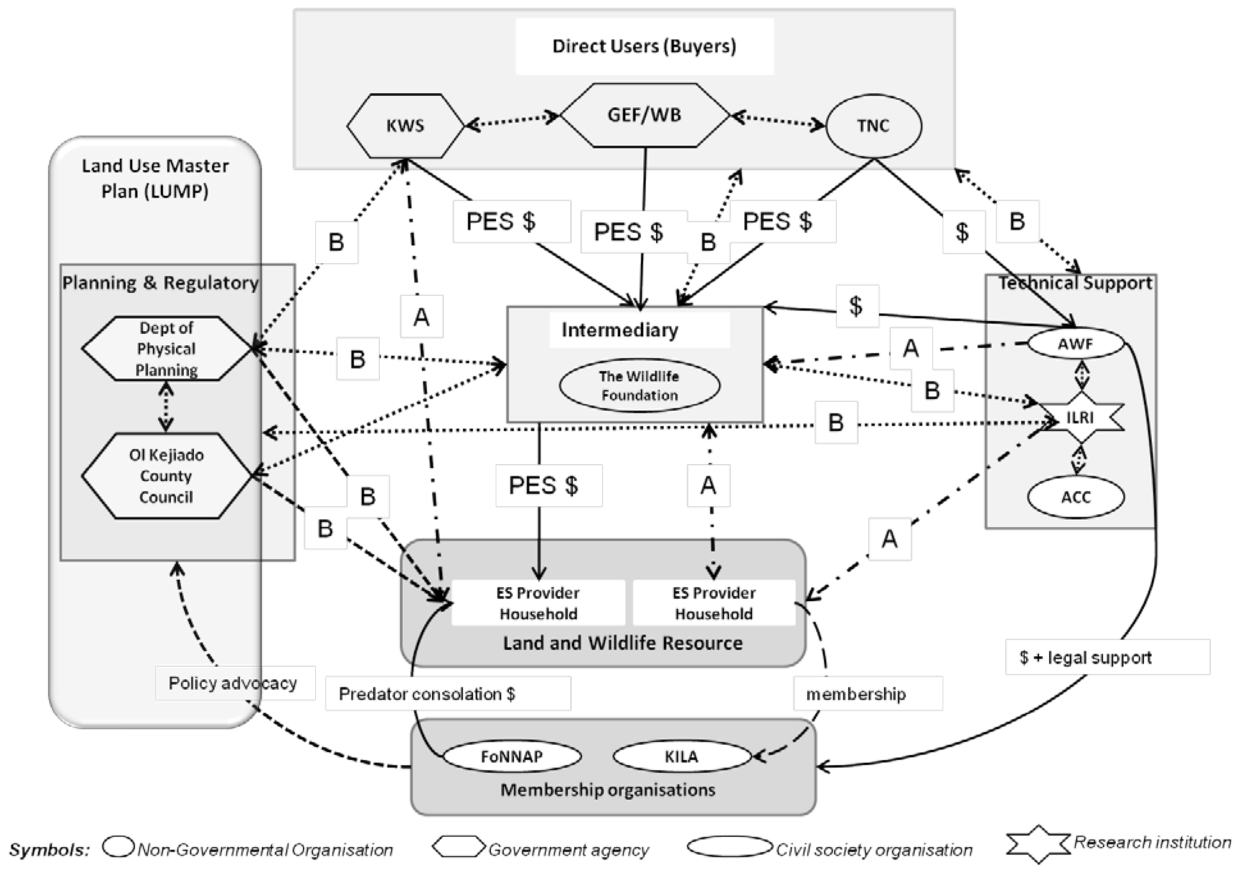
A total of 31 households dropped from the WLP (voluntarily or because of violations), representing an annual dropout rate of 3.1%. The majority of the current participants (92%) would like to remain in the WLP in the next five years, but more than half (58%) are not willing to accept legally binding long-term contracts such as land easements (Table 5.4).

On the demand side, TNC (The Nature Conservancy) withdrew and discontinued funding in 2010. The main reason for this discontinuation was because TNC undertook an African-wide prioritization and the Nairobi National Park Ecosystem which is encompassed in the Athi-Kaputie Plains did not emerge as a priority area of focus. In addition, TNC also reported to have been dissatisfied with the participants' selection criteria which was based on a first come-first served basis and usually not defined on ecological priorities. The Global Environment Facility (GEF)/World Bank funding support to the WLP ended in 2012 following the expiry of the grant provided to pilot the scheme. GEF grants are not renewable so it is unlikely that additional funding for the WLP will be obtained through the GEF/World Bank window. There is currently also uncertainty regarding future funding from the Kenya Wildlife Service (KWS) because no grant renewal has been obtained (as at January 2013). . While TWF is in negotiation with KWS to extend their funding grant, they have also established a process to seek funding support from other sources, including the local private sector.

Actors and stakeholders

Figure 5.3 present the schematic representation of the key actors and other stakeholders in the WLP and the institutional arrangements that currently exist. This diagram shows the complexity involved in the implementation of a PES scheme, with actors and stakeholders that range from local to international organizations in the civil society, government and research institutions.

Figure 5.3 A simple schematic representation of the structure and institutional arrangements of the Wildlife Lease Program (WLP).
Source; Developed by the lead author based on interviews with TWF and document analysis.
Acronyms: AWF – Africa Wildlife Foundation; ACC – Africa Conservation Center; ILRI – International Livestock Research Institute; KWS – Kenya Wildlife Services; GEF/WB – Global Environment Facility/World Bank; TNC – The Nature Conservancy; KILA – Kitengela Il Parakuo Land Owners Association; FoNNAP – Friends of Nairobi National Park
*Description of symbols***: A**: Assessment and monitoring; **B:** Consultation and Collaboration



Ecosystem service provider households consist of pastoral landowners in AKP. The intermediaries include the Friends of Nairobi National Park (FoNNAP) which piloted the WLP until 2002 and TWF the current implementing institution responsible for collecting money from the funders, enrolling and paying providers, and monitoring compliance. Three buyers fund the land leases; the KWS paid US$ 64,300 during 2007-2011 (US$ 12,860 per year); the GEF/World Bank paid US$ 792,000 during 2008-2012 (US$ 66,000 per year), and TNC disbursed US$ 150,000 during 2007-2010 (US$ 37,500/year).

Other stakeholders include a research institution (ILRI), local government (Ol Kejuado County Council), the central government (Ministry of Lands through the Department of Physical Planning and Surveys) and non-governmental organizations such as the Africa Wildlife Foundation (AWF) and the Africa Conservation Center (ACC). All these provide technical, legal and scientific support for the WLP, and are also involved in the development of a Land Use Master Plan (LUMP: Figure 5.3).

Ecosystem services

The WLP pays landowners per unit area of open rangeland enrolled, taken as a proxy for ecosystem service delivered. Thus, WLP can be classified as paying for 'habitat services' (TEEB, 2010) because the open rangeland supports wildlife dispersal, migration of large herbivores and also provides 'habitat' services for lifecycle maintenance and gene pool protection for the wildebeest which breeds in Triangle II of the AKP (Figure 5.1).

Conditionality and monitoring

As shown in Table 5.5 monitoring is currently undertaken for five of the six conditionality's focusing on three indicators: change in land area under fences; changes in population of wildlife species; and sub-division or sale of enrolled land.

Table 5.5 Conditionality, Indicators and Monitoring in the Wildlife Lease Program in Athi-Kaputiei Plains
Source: lead authors' survey.
Acronyms: **ILRI***:* International Livestock Research Institute; **DRSRS**: Department of Resource Surveys & Remote Sensing.

WLP Conditionality[a]	Indicator for monitoring	Monitoring, scale and frequency	Who monitors
To leave the land under the lease open and not to install any perimeter fencing on the land	Land area under fences	Yes; landscape level; irregular, *ad-hoc* basis	ILRI
Not to cultivate, mine, quarry in any manner on	Presence of mines, quarry and	Yes; landscape level; irregular,	ILRI

the land in the wildlife lease scheme	cultivated crops	*ad-hoc* basis	DRSRS
To actively protect wildlife and prevent poaching including picking up snares	Poaching incidences reported	Yes; irregular, *ad-hoc* basis	Community scouts
	Number of snares picked	Yes; irregular, *ad-hoc* basis	Community scouts
	Population of wildlife species	Yes; landscape level; Aerial and ground wildlife surveys/counts	ILRI DRSRS
To keep the land under the lease free from buildings and any other structures	Presence of buildings and other structures on enrolled Plots	Yes; plot level; regularity not determined	Community scouts
No sale and/or sub-division of leased land	Land ownership transfer based on change of title deed	Yes; plot level; regular	TWF
To protect indigenous plants and trees and plant indigenous trees on the land where appropriate	Indigenous trees protected and planted on plot	No evidence of monitoring	

[a]Conditionality as stipulated in the lease contract between TWF and the participating land owners

In terms of fences, the WLP was meant to ensure that the threat of rangeland fragmentation through fencing is minimized by providing incentives to landowners to avoid fencing their enrolled land parcels. There were no baseline data on the distribution and the area covered by fences in 2000 at the start of the WLP and the GEF/World Bank Project Implementation Plan for the WLP used a baseline of 10% of the WLP target area as enclosed by fencing but does not provide the actual number of fenced properties. The fence data for AKP, which provided the baseline for the WLP, were first collected in June–October 2004 by local Maasai trained and guided by ILRI researchers to map land uses using GPS technology in Triangle I and II of the AKP and the fence data was updated in 2009 when Triangle III was also mapped, again led by ILRI working with trained members of the local community.

An analysis of the fence data shows that in 2004, a total of 6,471 properties in Triangle I and II of the AKP were under fences (Reid et al., 2008) and a total of 10,267 properties were fenced in Triangle I and II combined (Said et al., Submitted). This shows an increase of 58% in the number of fenced properties in Triangles I and II between 2004 and 2009, although this data does not show whether the fenced properties are enrolled in the WLP or not. Overall, about 20% of the AKP landscape (Triangles I, II and III combined) was fenced by the end of 2009 (Said et al, submitted) (see Appendix II-A for 2004 and 2009 fence maps). Overall, by 2009, 20% of AKP was under fences (Said et al., Submitted).

In terms of wildlife populations, lions, leopards, wildebeest and zebra are taken as the indicator species (World Bank, 2008). The lion population in Kitengela which is located in Triangle I of the AKP is considered as a good indicator both of general wildlife populations (prey species) and level of tolerance of landowners to presence of wildlife on their land --and consequently an indication of the effectiveness of the combination of incentives and predation control measures supported by the WLP initiative.

Historically, there is no evidence of a sizeable resident lion population in the Kitengela. However, Kitengela serves as a dispersal area for the Nairobi National Park surplus lions and as a lion hunting reservoir in time of game scarcity in the Park, particularly during wet-season periods, when many of the grazing animals leave the Park. Consequently, the access to the Kitengela area is one of the important factors underlying the population stability and reproductive rates of lions in the NNP (Rudnai, 1979). There is no readily available comprehensive long term data on lion populations using the AKP. In 2011, 35 adult lions were recorded in NNP (FoNNAP, 2011) up from only 18 individual adult lions in 2003 showing that the lion populations have increased, although this cannot be attributed to the WLP *per se*.

The changes and distribution in the population of wildebeest and zebra which are the two most common herbivore species that disperse between the NNP and the adjacent lands in AKP (Imbahale et al., 2008) can also serve as a good indicator pointing to the effectiveness of the WLP. Recent studies on the dynamics of herbivores shows that it is only wildebeest populations but not zebra in NNP that have largely been affected by land use changes within the AKP leading to a collapse of the migratory wildebeest (Reid et al., 2008; Ogutu et al., submitted).

It is estimated that since 1997, the total wildebeest population in AKP is about 6,000, having fallen from almost 30,000 animals counted in 1978, while that for zebra is estimated at about 10,000 of which roughly 1,000-2,000 are within Nairobi National Park (Ogutu et al. Submitted). The AKP therefore remains a significant area for these two species, providing habitat to the majority of the 6,000 wildebeest (only 160-250 wildebeest were found in the park since 2000) and about 8,000 zebra (only 2000 use the park).

In terms of land sub-division and sales, the WLP managers are supposed to confirm, prior to payments, that there is no change in land ownership based on the title deed; if land is sold or sub-divided, a new deed is issued to show the change in ownership thus providing evidence of land sale or sub-division, but data was unavailable regarding the land sub-divisions among participants in the WLP.

Poverty and Inequality

Income and land poverty

The poverty rate, density and gap are lower in locations near the park and closer to Nairobi metropolis (rate< 18%; density<3 per Km^2 and gap<10%: Figure 5.4a; b; c). The highest poverty rate (48-55%: Figure 5.4a) and poverty gap (17-23; Figure 5.4c) occur to the west and south west of AKP while poverty density is higher in central and western part of AKP (Figure 5.4b).

The data on poverty gap shows that on average, per capita monthly cash transfers of KES 124 and KES 285 in areas with poverty gaps of 10% and 23% respectively will lift a poor person above the poverty line. The figures calculated by multiplying the poverty gap by the rural poverty line for 1999 of KES 1,239 (Central Bureau of Statistics, 2003).

Overall, 68% and 70% of all households were income-poor in 2008 and 2009 respectively, by Kenyan rural poverty standards (Earning less than KES 1,560/capita per month). Among participating households, 70% and 67% were income-poor in 2008 and 2009 respectively while among non-participants, 67% and 74% were poor in 2008 and 2009 respectively (Table 5.6).

The high prevalence of poverty still persist when income poverty is assessed based on the international measure of ‘extreme poverty’ of US$ 1 per person per day, such that 75% of all households were extremely poor in 2008 and 81% in 2009 (Table 5.6). Among participants, 74% and 78% were extremely poor in 2008 and 2009 respectively while for non-participating households, 76% and 85% were extremely-poor in 2008 and 2009 respectively (Table 5.6). Lastly, in terms of land ownership, 47% of all the surveyed households are ‘land poor’ (Table 5.6).

Figure 5.4 Map of the Athi-Kaputiei Plains showing the distribution of the surveyed households in relation to **A.** poverty rate; **B.** poverty density; **C.** poverty gap; and **D.** gini index
Data source; Kenya National Bureau of Statistics (KNBS) and lead author's survey

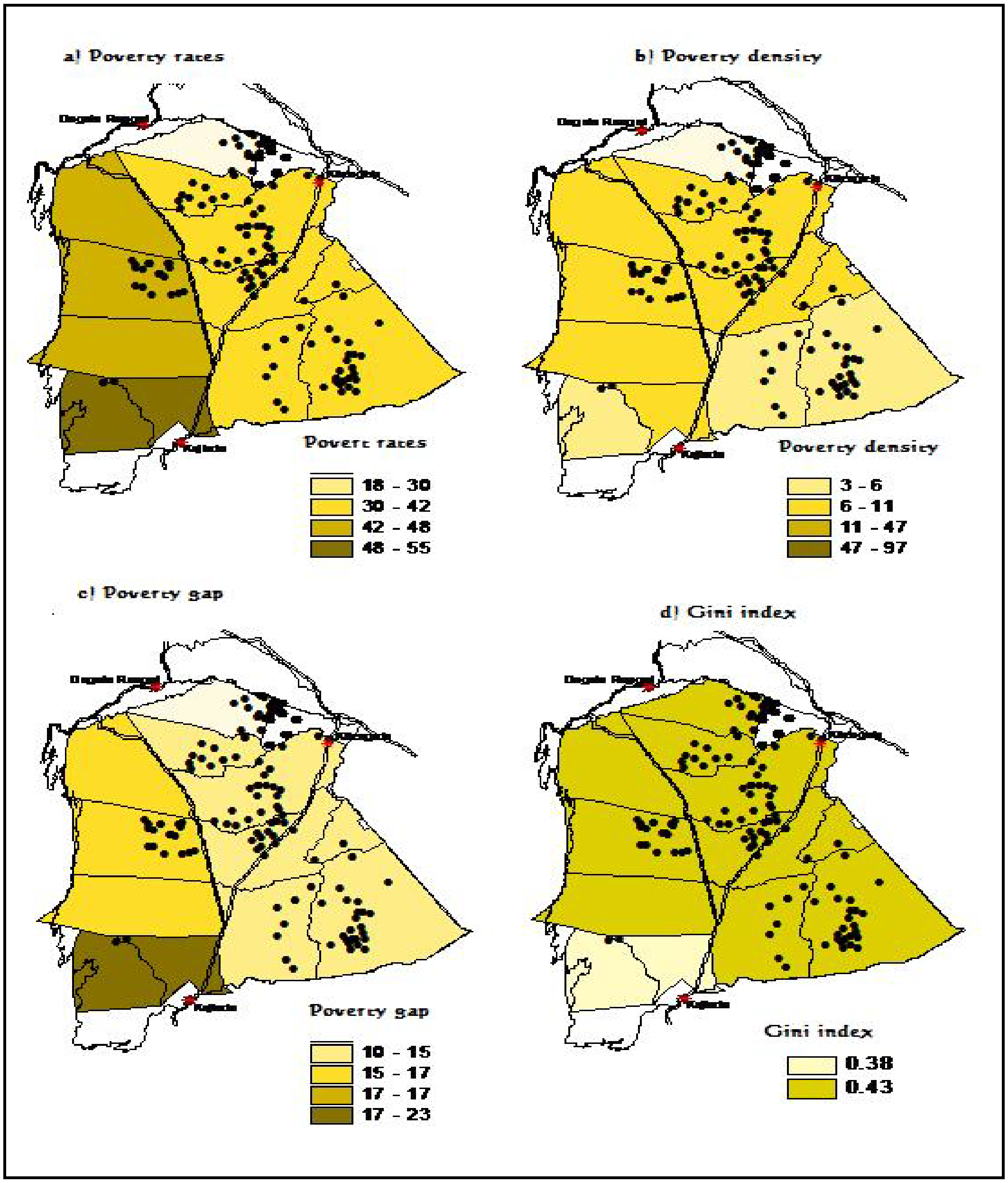


Table 5.6 Income and land poverty status of surveyed households in Athi-Kaputiei Plains in 2008 and 2009
Source; lead authors' survey

			2008							2009					
		Households (HHs)	All HHs		WLP[c]		Non-WLP			All HHs		WLP		Non-WLP	
			No	(%)	No	(%)	No	(%)		No	(%)	No	(%)	No	(%)
Income poverty	Kenya rural poverty line (KES 1,562/AE[a]/month)	Poor Households	108	(68)	60	(70)	48	(67)		111	(70)	58	(67)	53	(74)
		Non-Poor Households	50	(32)	26	(30)	24	(33)		47	(30)	28	(33)	19	(26)
		Total	*158*	*100*	*86*	*100*	*72*	*100*		*158*	*100*	*86*	*100*	*72*	*100*
	International poverty line (US$ 1/AE/day)[b]	Poor Households	119	(75)	64	(74)	55	(76)		128	(81)	67	(78)	61	(85)
		Non-Poor Households	39	(25)	22	(26)	17	(24)		30	(19)	19	(22)	11	(15)
		Total	*158*	*100*	*86*	*100*	*72*	*100*		*158*	*100*	*86*	*100*	*72*	*100*
Land poverty	Land poverty line is 80.94ha	Poor Households								74	(47)	37	(47)	37	(51)
		Non-Poor Households								84	(53)	49	(57)	35	(49)
		Total								*158*	*100*	*86*	*100*	*72*	*100*

[a]The concept of adult equivalent (AE) is based on differences in human nutrition requirements according to age, where; <4, 5-14 and > 15 years of age are equivalent to 0.24, 0.65 and 1 AE respectively.
[b]The conversion rates used are US$ 1=KES 63.20 (June 30, 2008) and US$ 1=KES 73.98 (June 30, 2009). Source: www.oanda.com
[c]WLP: Wildlife Lease Program

Inequality in income, land and livestock assets

All surveyed households with the exception of two, are found in areas recording gini-coefficient of 0.43 in terms of per capita expenditure (Figure 5.4d) suggesting relatively high inequity. Amongst all households surveyed the highest inequality is found in cattle ownership (gini index [GI] = 0.58), followed, in decreasing order, by land ownership (GI= 0.52), cash income (GI= 0.51) and sheep and goats (shoats: GI= 0.47), and significant difference between participants and non-participants only existed for income inequality (Table 5.7). We tested and found a weak correlation between land and income (r = 0.152) and between land and livestock (r= 0.202), but as expected, high correlation among the livestock assets (Appendix II-B).

Table 5.7 The Gini-index for gross cash income in 2009, land (among surveyed households and all households enrolled in the WLP in 2010), cattle, sheep and goat ownership in Athi-Kaputiei Plains and the differences in gini-coefficients between participants and non-participants. The test of significance reported in the last two columns.
Source; Lead authors' survey; The Wildlife Foundation (TWF) for WLP land ownership data

	Group	**Estimate**	**SE[a]**	**95% LCL[b]**	**95% UCL[c]**	**DIG[d]**	
							P>|t|
Gross Income in 2009	Non-participants	0.574	0.057	0.461	0.687	(1.942)	0.056**
	Participants	0.448	0.031	0.387	0.509		
	All	0.511	0.030	0.451	0.571		
Land ownership (Surveyed HHs)	Non-participants	0.477	0.041	0.395	0.559	(-0.863)	0.391
	Participants	0.523	0.034	0.456	0.592		
	All	0.516	0.029	0.459	0.574		
Land ownership (All WLP HHs)		0.515	0.020	0.476	0.555		
Cattle ownership (TLU)	Non-participants	0.625	0.049	0.529	0.721	(1.258)	0.213
	Participants	0.547	0.038	0.471	0.623		
	All	0.584	0.031	0.523	0.644		
Shoats ownership (TLU)	Non-participants	0.501	0.035	0.431	0.569	(1.166)	0.248
	Participants	0.448	0.029	0.392	0.505		
	All	0.473	0.022	0.429	0.517		

**P*< 0.1; ***P*<0.05; ****P*<0.01
[a]SE: Standard Error; [b]LCL: Lower Confidence Limit; [c]UCL: Upper Confidence Limit: [d]Differences in Gini-coefficient (degrees of freedom [DF] =71)

Determinants of participation and participation intensity

The probability of participation in the WLP increased significantly with increasing current plot area (farm size); wildlife predation; adult labor and composite asset index (CAI), but decreased with increasing NDVI (Table 5.8). Regression diagnostics based on the link test indicated no model mis-specification errors (z =-1.19, p=0.233) (Appendix II-C), and the Hosmer-Lemeshow's goodness-of-fit test (p= 0.1), indicated a good fit to the data (Appendix II-D). Furthermore, the estimated variance inflation factor (VIF) of 1.40 (Appendix II-E), is within acceptable level for logistic models, indicating structural adequacy of our model (Pregibon, 1981).

Table 5.8 Determinants of participation (n=158 households) in the Wildlife Lease Program (WLP) in Athi-Kaputiei Plains (Y=1 if the pastoral household participates, otherwise, Y=0) based on the logistic regression model.
The model explained 17% of the variance (χ^2_{13} =13) =37.36, Pseudo-r^2 =0.1716).
Data source; Lead authors' survey

Variable	**Coefficient**	**z**	**P>|z|**
Gender	-0.540	-1.22	0.222
Education	-0.264	-0.62	0.534
Current plot (ha)	0.005	2.51	0.012**
Employment status	0.373	0.96	0.338
Access to credit	-0.412	-1.02	0.307
Wildlife predation	0.863	2.05	0.041**
Total cash income (2009)	1.41×10^{-6}	0.71	0.480
Distance to town(km)	-0.026	-0.38	0.707
Total adult labor	0.163	2.04	0.042**
Child dependency ratio	0.114	0.55	0.582
Distance to primary school	0.020	0.19	0.848
NDVI_lag3	-9.769	-2.33	0.020**
Composite Asset Index (CAI)	0.622	1.82	0.069**
Intercept	-0.747	-0.57	0.571

**P*< 0.1; ***P*<0.05; ****P*<0.01

The intensity of participation in the WLP increased significantly with increasing adult labor and current plot area (farm size) (Table 5.9).

Table 5.9 Determinants of intensity of participation (n=86 participants) in the Wildlife Lease Program (WLP) in Athi-Kaputie Plains (AKPs) during 2000-2010 period. NDF and DDF are the numerator and denominator degrees of freedom respectively.
Data source: Lead authors' survey

Effect	**Estimate**	**Std. Err**	**NDF**	**DDF**	**t**	**P>|t|**	**F-Test**	
							F	P> F
Intercept	*-0.92*	*0.76*	*1.00*	*149.00*	*-1.21*	*0.23*		
Total adult labor	0.10	0.06	1.00	149.00	1.59	0.11*	4.557	0.034
Child dependency ratio	-0.10	0.21	1.00	149.00	-0.46	0.65	0.124	0.725
Total cash income (2009)	0.00	0.00	1.00	149.00	0.42	0.68	0.813	0.369
Distance to nearest town	0.03	0.07	1.00	149.00	0.38	0.70	0.009	0.925
Distance to nearest road	0.09	0.09	1.00	149.00	1.01	0.31	0.833	0.363
Distance to nearest primary school	0.01	0.10	1.00	149.00	0.11	0.92	0.114	0.736
NDVI_lag3	-3.85	3.72	1.00	149.00	-1.03	0.30	0.385	0.536
Current plot (ha)	0.00	0.00	1.00	149.00	2.23	0.03**	4.961	0.027

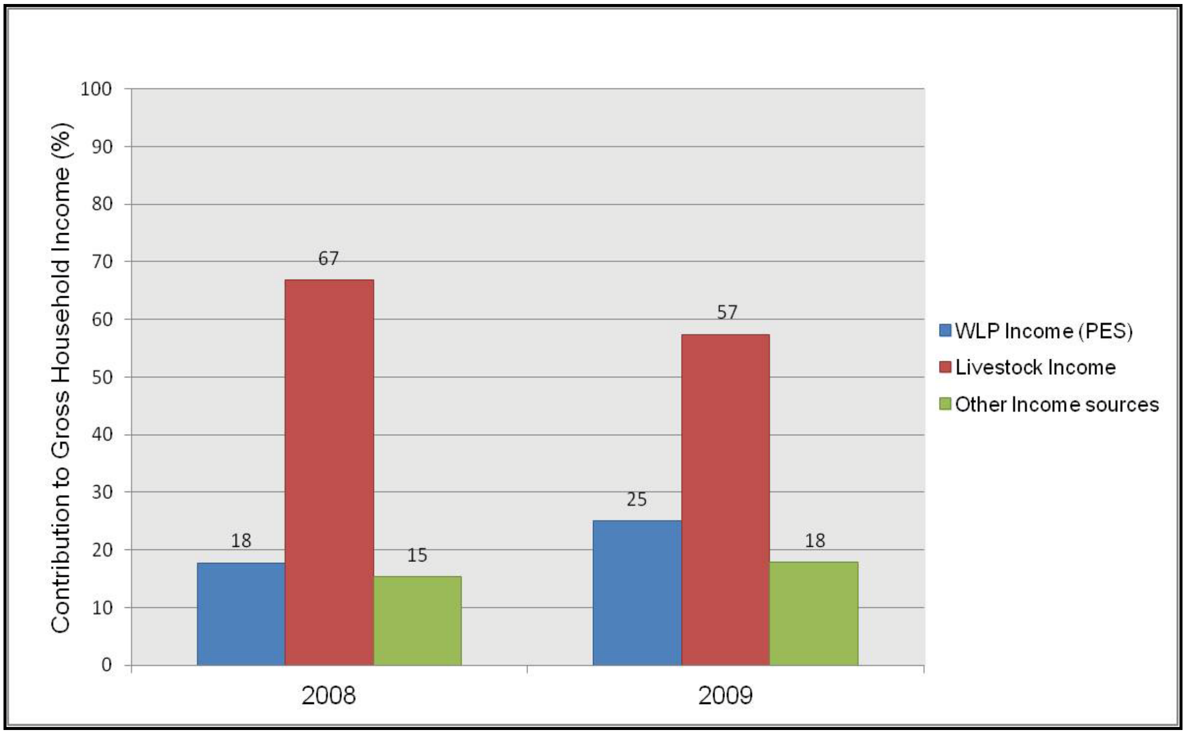
*$P<0.1$; **$P<0.05$; ***$P<0.01$.

The examination of raw, studentized and Pearson residuals indicated reasonable model fit (Table 5.9)

Financial income impacts on participating households

Participating households received a mean annual PES income of US$ 328 and US$ 345 in 2008 and 2009 respectively (Table 5.10). This represented 18% and 25% of the gross household cash income (Figure 5.5). Although livestock income represented the largest share of total gross household cash income, it declined by 10% point between 2008 and 2009 while the share of gross household income accounted for by PES increased by 7% points over the same period, and was higher than that of other income sources combined in both years (Figure 5.5). This suggests that the WLP PES payment serves as an invaluable source of income diversification for participating households especially during periods of drought when cash income obtained from livestock may decline as the case in 2008 and 2009 period.

Figure 5.5 Contribution of the Wildlife Lease Program (WLP) PES, livestock and other income sources to the gross annual household income in 2008 (n=61) and 2009 (n=86). *Data source*: Lead authors' survey.



Moreover, the WLP PES income is the most equitably distributed across households as it had the lowest percentage coefficient of variation in both 2008 (%CV=74) and 2009 (%CV=109) (Table 5.10).

Table 5.10 Cash income and sources for participating households in 2008 (n=61)[a] and 2009 (n=86).
Data source: Lead author's survey.

Income source	Year	Mean (US$/yr)[b]	Households reporting income		SD	Median	Max	Min	CV%
			No.	(%)					
Livestock sales	2008	1,240	53	(87)	1,253.74	1,000	6,429	0	101
	2009	800	76	(88)	992.32	536	5,000	0	124
Salaries/ Wages	2008	236	23	(38)	606.61	-	4,286	0	257
	2009	202	34	(40)	521.09	0	4,286	0	258
Remittances	2008	35	8	(13)	111.78	-	571	0	318
	2009	36	10	(12)	120.60	0	614	0	338

Wildlife (WLP)	2008	329	61	(100)	241.87	257.24	1,429	40	74
	2009[c]	345	86	(100)	377.65	221	2,421	0	109
	2008	14	1	(2)	109.75	-	857	0	781
Crop	2009	10	3	(3)	77.48	0	714	0	781
Other Income	2008	<1	1	(2)	2.56	-	20	0	927
	2009	1	2	(2)	7.97	0	71	0	750
Gross Income	**2008**	**1,546**	**61**	**100**	**1,251.12**	**1,668.57**	**6,557**	**40**	**67**
	2009	**1,394**	**86**	**100**	**1,235.97**	**1085**	**6,371**	**19**	**89**

[a]Twenty five (25) households that participated in the survey joined the WLP in January 2009 and did not receive payment from the WLP.
[b]Calculated on exchange rate of US$= KES 70
[c]One household joined the WLP in January 2010 and therefore did not receive income from the WLP in 2009

Discussion

Design and implementation of the WLP

Our analysis reveals four critical issues. First, the WLP is perceived to be effective if the participating landowners maintain their current land use practices of livestock grazing while keeping open rangelands and not fencing, selling or sub-dividing enrolled land. It is therefore unsurprising that the WLP has very low default rate and is oversubscribed despite offering uniform payments. This point to potential adverse selection; some households are probably underpaid while others overpaid relative to their opportunity cost.

Enrolment in the WLP is not targeted either for biodiversity or poverty outcomes, but rather is based on a 'first-come' basis. Although uniform payments are common in publicly funded PES schemes (Wunder et al., 2008), for the WLP, it is also justified by the need to maintain community cohesion. The WLP administrators at the TWF pointed out that local landowners might perceive differential payments as 'unfair' leading to dissatisfaction in the WLP but this is despite the fact that some participating landowners living closer to the park would like to be paid a higher rate than landowners further away from the park.[3] However experience with biodiversity PES schemes shows that

[3] This view was presented by a community leader during an appraisal of the WLP by a team from the World Bank on September 17, 2009

uniform payments are often not cost-effective because of the failure to account for heterogeneity in the delivery of ecosystem services and the spatial variation in opportunity costs (OECD, 2010, Wünscher et al., 2008).

Second, it is evident that the involvement of other actors in addition to the intermediaries and buyers is critical in the provision of the relevant technical, scientific and legal expertise necessary for PES' implementation. This points to the need for PES designers to take cognizance of the multiple actors and interactions that span different sectors and scales (Reid et al., 2009).

In terms of funding, the WLP is a hybrid model that derive finances from both public and private sources (Engel et al., 2008). The GEF/World Bank and TNC operate globally and can be considered as paying for non-use values of wildlife in AKP that accrues to the global community (Farley and Costanza, 2010). On the other hand, KWS is a Kenyan government institution with the statutory mandate to protect wildlife and manage all national parks, including the NNP for which it collects visitor fees. Thus, KWS financing of the WLP can be considered as either constituting public funding (Wunder et al., 2008), or as indirect payment by visitors to NNP for the non-consumptive use values of the wildlife (Navrud and Mungatana, 1994) which uses both the park, and the private land in the AKP.

The sustainability of the current WLP funding is however, not guaranteed as the GEF/World Bank financing ends in 2012 (World Bank, 2008) and continued financing by KWS remains uncertain. Although financing challenges are not uncommon in government-funded PES schemes in the developing world (Wunder et al., 2008), there is need to explore alternative sources of future financing, including options such as increasing NNP user fees, or charging a nominal fee to beneficiaries of NNP's ecosystem goods and services, and channeling the additional revenue generated into the WLP (Rodriguez et al., 2011a).

Third, by limiting fragmentation, the WLP has contributed to controlling land use changes such as cropping which negatively affect wildlife in private land outside the NNP but also supports pastoralism (Reid et al., 2008). Since protected areas are ecologically linked to their surrounding lands (Hansen and DeFries, 2007) and collectively provide ecosystems services which are essential for human sustenance (DeFries et al., 2007), the WLP plays a critical role in managing land outside NNP to balance both ecological and human needs. However, compliance with the Conditionality imposed by PES appears to be a challenge. Controlling fencing will need to be strengthened if the WLP is to be effective. The research findings with regard to fencing further highlight several issues worth consideration. First, pastoral households generally do not fence their grazing lands. A 1999 survey revealed that nearly all households with "boma" (homestead) closer to the Nairobi National Park in Triangle I have fences around their homes; 83% of them around small cultivated plots next to their homes, and only 16% have fences around any of their grazing lands (Reid et al., 2008). Secondly, analysis by Said and colleagues found that the major roads had a huge influence on the location of fences. About 75% (10,700 out of more than 15,000) of fenced land parcels were located along the major roads which traverse the migratory routes of wildebeest and zebra between their wet and dry season ranges (Said et al. submitted).

Lastly, from an ecological perspective, the functional value of the entire wildlife dispersal area in AKP, is realized at the landscape level because of the need to manage larger areas of land to ensure connectivity that allows flexible migration of animals (Rouget et al., 2006, Sanjayan and Crooks, 2005) but PES payments are provided at the plot level and are based on outcomes and not results. This leads to disconnect between the scale at which payments are made (plot level based on action), and the scale at which results are anticipated (landscape level). We suggest that PES practitioners need to pay careful attention to scale in designing biodiversity PES schemes (OECD, 2010) and consider paying for results rather than outcomes (Gibbons et al.).

The implication of scale mismatch is evident where the land enrolled in the WLP (an outcome) has increased but alongside an increase in the area under fences (Said et al., Submitted) yet WLP pays to restrict fencing. Even if the increases in fenced areas occurred in non-enrolled land, these could still undermine the effectiveness of the WLP at the landscape level. Also, not all land parcels required to ensure positive WLP impact on wildlife are currently enrolled and this may create the challenge of ‘hold-outs’ (Salzman, 2005). This can occur for example, if there are two contiguous land parcels and only one of the two parcels is enrolled in the PES scheme while the other is not. If the owner of the non-enrolled parcel decides to put up a fence, then this action will effectively undermine the efforts of the owner of the enrolled parcel to avoid fencing. This situation will undermine the effectiveness of the WLP. In addition, there are also challenges arising from collective action decision and the complexity involved in bringing together the many contiguous land parcels to a scale sufficient for effective impact of a PES scheme at the landscape level (Swallow and Meinzen-Dick, 2009).

Poverty and Inequality

The high poverty rates in parts of the AKP are an indicator that potential participants are also likely to be poor. The poverty densities are generally low because of low population density in AKP, and the areas with the highest depth of poverty record gap of 23%. Our analyses also show that three-quarters of all the surveyed households are income-poor by national and international standards, and half of them can also be considered as land-poor. Our results support similar findings from earlier studies covering AKP (Kristjanson et al., 2005, Kristjanson et al., 2002). For example, 80% of households in Triangle I were classified as income poor; 54% ‘extremely poor’ and 26% as ‘poor’ earning less than one or two US$ per day per AE respectively (Kristjanson et al., 2002).

Despite our findings showing high levels of household income poverty, the latest government report on poverty at the County level shows that Kajiado County, within which AKP is located, recorded the highest reduction in poverty rate (incidence) among all the 47 Counties in Kenya, from 44% in 1999 (Central Bureau of Statistics, 2003) to

11.6% in 2005-2006 (Commission on Revenue Allocation (CRA), 2011).[4] This suggests potential heterogeneity in poverty incidence at the local level, affecting pastoral households which are masked by average values at the County level.

The implications of these findings on poverty are varied. First, the correlation between the location of potential participants and high poverty rate in AKP shows the potential of the WLP to involve poor households as participants (Pagiola et al., 2005, Grieg-Gran et al., 2005). Second, the poverty gap, which measures how far on average, the poor are below the poverty line, can provide an indication of whether PES cash transfers can lift the poor above the poverty line. The poverty gap ranges from a low of 10% to a high of 23%, meaning that, on average, a poor person will require an additional monthly income of KES 123 and KES 285, to move above the rural poverty line of KES 1,239 in locations with poverty gaps of 10% and 23% respectively. The WLP provide an annual payment of KES 741 per hectare, which translates to a monthly income of KES 62 per hectare. The mean land enrolment among the 357 households in the WLP is 87ha, so the WLP transfers, on average, a monthly per capita cash income of KES 5,403, which is of a magnitude sufficient to lift all households found in locations with a poverty gap of 23% and below, to the rural poverty line.

The poverty gap however should be considered in light of the opportunity costs involved for the participating landowners, but which have not been estimated. Furthermore, the poverty gaps represents an average which conceals the variation in the depth of poverty, so it only provides a crude indication of PES impact on poverty based on the magnitude of household cash transfers, which may be useful in linking PES to poverty, for example through “Payments for Ecosystem Services and Poverty Alleviation (PESPA)” programs (Rodriguez et al., 2011b).

Income poverty is an absolute measure, but poverty is also relative (Coudouel et al., 2002) and multi-dimensional (World Bank, 2001). Thus, we also consider land

[4] The data on poverty rates by Districts in Kenya is publicly available at the Government of Kenya Open Data website online at http://opendata.go.ke/Poverty/District-Poverty-Data-KIHBS-2005-6/pnvr-waq2

ownership, a relative poverty measure, which is both relevant to pastoralist perception of poverty (Tache and Sjaastad, 2010) and is an asset-based poverty indicator (Reardon and Vosti, 1995). It has been observed that high levels of inequality between PES participants and non-participants in terms of land ownership and income suggests that PES may perpetuate existing disparities in wealth (Leimona and Joshi, 2009). Our results also demonstrate that while PES does reduce income inequality among participants, as has also been found in the publicly funded Sloping Land Conversion Program (SLCP) PES scheme in China (Li et al., 2011). PES however does increase the income gap between participating and non-participating households given the significant cash income difference between these two groups.

Overall, the high levels of inequality in income, livestock and land among households in AKP are not unusual (Rutten, 1992, Radeny et al., 2007). The gini-index for income for Kenya is 0.47 (2005: www.worldbank.org) which reflects high inequality in the country, and AKP lies within a constituency in Kajiado County that has the highest inequality in per capita expenditure in Kenya (Central Bureau of Statistics, 2005). The coefficient of variation (CV) for cattle and land ownership based on a survey of 220 households in Kajiado County is estimated at 276% and 157% respectively (Kabubo-Mariara, 2005).

Determinants of participation

Farm size is positively and significantly associated with participation, suggesting that an increase in total land increases the probability of participation in the WLP thus implying a negative poverty effect (Wunder, 2008, Grieg-Gran et al., 2005). Farm size is a critical factor in many PES programs, and is a significant determinant of participation in the PSA PES scheme in Costa Rica (Zbinden and Lee, 2005) and among households selected for participation in the Chinese Sloping Land Conversion Programme (SLCP (Mullan and Kontoleon, 2009).

Increased intensity of wildlife predation on livestock also increases the probability of participation in the WLP. Although there is a significant difference between WLP participants and non-participants in terms of wildlife predation, our finding provides empirical evidence of the effects of the WLP payments on human-wildlife interactions, and identifies potential synergies between the WLP, and a separate 'Wildlife Consolation' scheme that compensates households in AKP for livestock predation to avoid retributive killings of carnivores (Sikand, 2007).

In effect, the WLP PES serves a dual function in this case. It is both an incentive for conserving private land for wildlife use (Pagiola, 2003) and a tool for mitigating human-carnivore conflicts (Dickman et al., 2011, Nelson, 2009), leading to a positive attitude towards wildlife among participating landowners (Nkedianye, 2004). This is not trivial as livestock depredation by wildlife is a serious challenge to millions of livestock keepers living adjacent to wildlife protected areas worldwide and can cause average annual losses of more than two-thirds of household financial income (Dickman et al., 2011). PES can be invaluable in ameliorating poverty caused by wildlife predation (Ura et al., 2009).

The increase in the probability of participation in the WLP with increasing adult labor, suggests that households with more adults of working age are more likely to shift adult labor to off-farm employment, implying pastoral household diversification (Homewood et al., 2009b). Currently, the effect of PES on family labor (or vice-versa) is unclear. Our results is consistent with studies such as on the *Grain for Green* PES program in China which shows that, on average, PES has a positive but moderate effect on off-farm labor participation, with PES participants allocating family labor into the off-farm labor market (Uchida et al., 2009), and a separate study that also found weak evidence that participating households in the SLCP have began to shift their labor to non-farm sectors (Uchida et al., 2007).

Our result is however inconsistent with PES studies in China and Costa Rica which found that participation in a PES program is not associated with greater transfer of labor toward non-farming activities (Li et al., 2011) and is negatively associated with greater availability of labor (Zbinden and Lee, 2005) respectively. Nevertheless, we expect the effect of PES on labor to be different between “use-restricting” (land-diversion) PES such as the WLP and “asset-building” (working-land) schemes such as the Costa Rican PSA program because of their different labor intensity demands (Zilberman et al., 2008, Wunder, 2008).

The negative correlation between the index of pasture potential (NDVI) and participation indicates that households located in areas with lower pasture/vegetation potential have a higher probability of participating in the WLP. As NDVI is negatively correlated with poverty at the community level in AKP (Kristjanson et al., 2005), our result suggests a positive poverty effect of the WLP as landowners in areas with less green vegetation and poor pasture quality (and also associated with high poverty rates) are more likely to participate in the WLP. This reinforces the proposition that where the sellers of environmental services are diverse in terms of farm size, environmental quality, and wealth, land diversion PES programs are only likely to benefit the poor when they are owners of lands that are poor for agricultural production but high in environmental service potential (Zilberman et al., 2008)

The probability of household participation also increased with increase in the coefficient for the composite asset index (CAI). This positive association suggests that wealthier households, with more valuable assets, have a higher probability of participating in the WLP than their poor and asset deprived counterparts, implying a negative effect on poverty. The remaining eight predictors are not significant but all had the expected signs.

Gender exerted a negative effect, implying that female-headed households have a higher probability of participating in the WLP than male-headed households. Similarly, education has a negative influence, suggesting that household heads with low education have a higher probability of participating in the WLP than those with better education. Employment has a positive coefficient, suggesting that employed household heads are more likely to participate in the WLP compared to their unemployed counterparts. This reinforces our finding that adult labor positively influences the likelihood of household participation.

As expected, access to credit exerts a negative impact on participation. Households without access to credit facilities are less likely to participate than those that have. Distance to the nearest town exerts a negative impact on participation, suggesting that households with land nearest to towns have a higher probability of participation in the WLP than households with land farther from towns. To the contrary, distance to primary schools has a positive coefficient, implying that households with land farther from schools are more likely to participate than those closer to schools. Child dependency exerts a positive impact on participation suggesting that households with more dependent children are more likely to participate than those with fewer. Income, though not significant, had a positive coefficient implying that households with high income levels have a higher probability of participation than household with lower income levels.

Determinants of the intensity of participation

Only Farm size and adult labor are positively and significantly associated with the intensity of participation in the WLP, and both of these variables are also significantly and positively associated with participation in the WLP. An increase in farm size also increases the proportion of land that a participating household allocates to the WLP, implying a negative PES effect on poverty as large landowners are likely to have a higher intensity of participation than small landowners. This observation has also been made in PES schemes in Colombia (Pagiola et al., 2010) and Nicaragua (Pagiola et al., 2008) where farm size was also found to be positively, and significantly associated with the intensity of participation in PES programs.

Our result suggests that households with more adults of working age allocate a higher proportion of land to the WLP, which again, imply a diversification effect. The effect of family labor on the intensity of participation in PES schemes is mixed. On the one hand, family labor is found to be negatively and non-significantly associated with the intensity of participation in a PES scheme in Nicaragua (Pagiola et al., 2008). On another hand, it was found to be positively and non-significantly associated with the intensity of participation in a PES scheme in Colombia (Pagiola et al., 2010).

All the spatial variables (distance to town, road and primary school) are positively associated with the intensity of participation in the WLP but are non-significant as is income. The child dependency ratio and the NDVI, although non-significant, are however negatively associated with the intensity of participation. The former suggest again that households with land in areas with poor vegetation/ low pasture quality (and are likely to be poor) have a higher probability of allocating a higher proportion of their land to the WLP, implying a positive poverty effect.

Financial income effects of the WLP on participating households

Our analysis reveals three critical insights. First, PES is an invaluable source of income diversification providing the second highest contribution to household income after livestock. The WLP payments accounted for 25% of the households' gross income in 2009 up from 7% in 2004 (Nkedianye et al., 2009). This is relatively high compared to other PES schemes that range from 0.02-15% in Costa Rica (Miranda et al., 2003, Grieg-Gran et al., 2005, Kosoy et al., 2007), 0.4-1.2% in Honduras (Kosoy et al., 2007) and 1-2% in Vietnam (Wunder, 2008), for example.

Our figures are probably underestimates because we conducted our survey during a period of severe drought, the worst in AKP since 1982 (Zwaagstra et al., 2010) and during which pastoralists lost up to three-quarters of their livestock (Osano, 2011, Western, 2010). We report gross income but it has been pointed out that the magnitude of PES payment itself is not a suitable measure of the financial benefit to participants, and the appropriate measure should be the payment net the PES opportunity cost (Pagiola et al., 2005). We do not estimate the opportunity and transaction costs of the WLP but acknowledge their importance in terms of poverty effect of PES (Grieg-Gran et al., 2005, Wunder, 2008, Kosoy et al., 2008).

Second, AKP receives mean annual rainfall ranging from 500 mm in southeast to 800 mm in the northwest (Reid et al., 2008) and the most suitable agricultural land use, based on its agro-ecological potential, is extensive livestock production (Kristjanson et al., 2002). Accordingly, the annual WLP payment of US$ 10 per hectare is close to the recommended annual compensation rate for wildlife conservation of US$ 8.6 per hectare (Bojo, 1996), as well as the estimated net annual return from livestock production of US$ 7.2 per hectare in Triangle I (Kristjanson et al., 2002), and US$ 7.6 and US$ 13.1 for areas that receive 600 and 700 mm of annual rainfall respectively (Norton-Griffiths and Said, 2010).

The current WLP rate is however not competitive relative to the expected returns from other forms of commercial land use such as real estate (Norton-Griffiths, 1998). The high land value and the rising land prices in the study area reflects the peri-urban and urban potential of land where wildlife has no future, and not the agro-ecological potential of land, which allows for wildlife as a form of land use (Norton-Griffiths and Said, 2010). This issue is pertinent because we assume in our analysis that landowners' decisions to enroll in the WLP is based on the agro-potential capacity of the land (which allows them to continue grazing livestock), and not its peri-urban or urban potential. In reality, many landowners continue to sell land (ACC, 2005, Nkedianye et al., 2009), suggesting, in part, that their decisions are informed by an appreciation of the peri-urban and urban potential of the land (Norton-Griffiths and Said, 2010).

We do not currently know the extent of land sales in the area as land sale records are not publicly available but interviews with selected funders, the TWF and key informants revealed that the rising land prices, land sales, sub-division and fencing are perceived as the greatest threats to the WLP (Reid et al., 2008, Nkedianye, 2004, Sikand, 2007, Norton-Griffiths and Said, 2010). Land prices in parts of AKP, closer to the park, near tarmacked roads, and around other urban centers are as high as KES 2 million per hectare (US$ 28,570 at exchange rate of $=KES 70) (ACC, 2005) and can reach up to US$ 200,000-400,000 per hectare (Norton-Griffiths and Said, 2010) and are likely to further increase, fuelled by high population growth and rapid urbanization including the recent incorporation of the AKP within the Greater Nairobi Metropolis (Government of Kenya, 2008b).

The irony here is that the pastoral landowners in AKPs are considered poor in monetary terms, but they hold highly valuable land assets (Norton-Griffiths and Said, 2010), a classic manifestation of the ‘land-rich and money-poor’ phenomenon. Selling land may allow some to convert their physical capital (land) to financial capital (cash income) but may not necessarily be a pathway out of poverty (Nkedianye et al., 2009, Rutten, 1992). These complex dynamics involving land use, land prices and changing land values suggest the need to pay attention to the interactions between land markets and conservation land use in PES schemes closer to rapidly growing urban centers (Armsworth et al., 2006).

Although enrolment in the WLP bars a landowner from cropping or selling land, it still remains popular and is oversubscribed. This suggests that land owners’ enrolment may be driven by other non-financial motivations as the evidence suggests they are not seeking to maximize profit by putting their land to its most economically productive use (Norton-Griffiths and Said, 2010). A partial explanation lies in the fact that, first, the WLP allows landowners to retain control of their land against an increasing pressure of land alienation for expansion of conservation and wildlife tourism estate (Goldman, 2011), and second, it complements, rather than replaces pastoral livestock production (Victurine and Curtin, 2010), which remains the economic mainstay for the majority of the households. Pastoralism is a highly valued socio-cultural practice for the Maasai community (Seno and Shaw, 2002), suggesting that cultural values might be an important factor driving participation in the WLP. Qualitative studies of PES schemes in Mexico have also highlighted non-economic factors, including sacred values of forests and intergenerational equity as critical participation drivers (Kosoy et al., 2008).

Conclusion

Global environmental policy strongly encourages the expansion of payment for environmental services (PES) to support biodiversity conservation and poverty reduction in developing countries (TEEB, 2009, FAO, 2007). Wildlife conservation agencies are adopting PES as a tool for conservation on private land (Pagiola, 2003, Frost and Bond, 2008), and to ameliorate human-wildlife conflicts (Zabel and Engel, 2010, Dickman et al., 2011). There remains a deep concern and little empirical evidence of the poverty implications of PES, especially among pastoral communities (Bulte et al., 2008a, Zilberman et al., 2008) regarded as among the poorest in Africa.

Our study evaluated the poverty effects of the Wildlife Lease Program (WLP), a PES scheme that pays pastoral land users adjacent to Nairobi National Park in southern Kenya to refrain from cultivation, land sales and sub-division, and to allow wildlife on their private land. The institutional architecture of the program is very complex. Besides the funders, participants and intermediaries, the WLP also involved multiple actors, including research institutions, NGOs and local groups that contribute technically and scientifically in its implementation. Monitoring is only undertaken irregular and unsystematically.

Although the scheme provides uniform payments, it is oversubscribed and has low default rates suggesting potential adverse selection. The determinants of participation in the WLP comprise three broad factors; farm size, human capital in terms of labor availability, household economic factors, in terms of asset ownership, human-wildlife interactions based on risk exposure to livestock depredation, and pasture conditions. These findings suggests that participation in the WLP is not strictly pro-poor as despite a bias towards poor household occupying areas with lower grazing potential, generally, the non-poor households with greater asset endowments, including larger farms, are the most likely to participate, and if already enrolled, have the highest participation intensity.

PES income is the most equitable and thus lessens income inequality among participating households. However, it also has the effect of widening the income inequality between participating and non-participating households. Without accounting for the opportunity costs incurred, the magnitude of the PES payment is, on average, sufficient to lift the poor households above the rural poverty line. PES also serves as an invaluable source of cash income diversification which contributes a substantial proportion of household income in periods of drought.

The WLP has the added advantage of land use restrictions that are largely compatible with traditional pastoral livestock grazing practices. Our study provides critical insights regarding the implications of PES on poverty and inequality in pastoral settings and provides useful lessons to inform the implementation of PES in pursuit of policies such as the UN Millennium Development Goals that seek to integrate biodiversity conservation and poverty reduction (Sachs et al., 2009).

BRIDGE BETWEEN CHAPTERS 5 AND 6

Both Chapters 5 and 6 of the thesis analyze and present results of the institutional arrangements, the implementation and the impact of Payments for Ecosystem Services (PES) schemes on household poverty and inequality. Although focusing broadly on the same question of PES impact on poverty at two different sites, Chapters 5 and 6 provide an illuminating contrast regarding PES implementation in different contexts. These contrasts emerge in the following three areas. The first is with regard to funding sources. Chapter 5 is based on a publicly funded PES scheme while Chapter 6 is based on a self-organized privately funded PES scheme. The second is with regard to PES Conditionality. Chapter 5 involves a PES scheme that broadly complements traditional pastoral livestock grazing on land enrolled in PES while Chapter 6 involves a PES scheme which excludes to a large extent, pastoral livestock grazing on land enrolled in PES. Lastly is with regard to the prevailing land tenure regime. Chapter 5 is based in a setting where land privatization and sub-division occurred in the late 1980s and early 1990s, and the land is now fully sub-divided. Chapter 6 on the other hand is based in an area where land is currently in the process of being sub-divided to individual holdings with the land within the conservancy area completely sub-divided but some areas in the larger study area are yet to be sub-divided into individual plots.

Chapter 6 uses the data from the study site in Maasai Mara Ecosystem (MME). The analysis is based on a case study of a user (market) funded PES scheme in the Olare Orok Conservancy (OOC). In this PES arrangement the Maasai pastoral landowners have agreed to voluntary resettlement and exclusion of livestock grazing from their recently sub-divided land parcels which are set aside for wildlife tourism, in return for cash payments worth US$ 41/ha/year (2011) provided by a coalition of tourism operators. In Chapter 6, I review the changes in land tenure in the MME in the period between 1959 and 2005, then assess and present findings on the institutional arrangements and the impact of the OOC PES scheme on household poverty, wealth and inequality and expenditure.

CHAPTER 6: WHY KEEP LIONS INSTEAD OF LIVESTOCK? AN ASSESSMENT OF THE EFFECT OF WILDLIFE TOURISM- BASED PAYMENT FOR ECOSYSTEM SERVICES ON HERDERS IN THE MAASAI MARA ECOSYSTEM, KENYA

Philip M. Osano, Mohammed Y. Said, Jan de Leeuw, Nicholas Ndiwa, Dickson Kaelo, Sarah Schomers, Regina Birner, Joseph O. Ogutu

Abstract

This paper examines the effects of wildlife-based tourism implemented through Payments for Ecosystem Services (PES) on household poverty, wealth inequality and livelihoods under changing land tenure in the semi-arid Maasai Mara Ecosystem (MME), south-western Kenya. It uses the case of Olare Orok Conservancy (OOC) PES scheme where Maasai landowners have agreed to voluntary resettlement and exclusion of livestock grazing from their aggregated land parcels which are set aside for wildlife tourism, in return for cash payments by a coalition of private commercial tourism operators. The results suggests that land privatization appears to have provided households with the exclusive rights to land, enhancing security of tenure, thereby enabling their participation in the PES scheme through a direct payment model that guarantees a transparent and relatively equitable benefit sharing of wildlife tourism revenues. The PES scheme was found to generate both positive and negative impact on the livelihoods of both participants and non-participants. On the positive side, first the magnitude of the PES cash transfer not accounting for the opportunity costs, is on average, sufficient to close the poverty gap. Secondly, PES serves as an invaluable source of income diversification, providing regular cash income to participants that is mostly spent on purchasing basic needs such as food, and maintains and restocks livestock herds during periods of severe drought. Third, PES also has an income inequality reduction effect among the enrolled participants. On the negative side, the PES induced land use restrictions appear to undermine pastoralism, and could also potentially lead to leakages with negative implications in the pastoral commons and the Maasai Mara National Reserve. There is need to assess and mitigate the PES impact on the non-participants and the landless and the potential leakages resulting from displacement of livestock and settlements and to better integrate tourism and wildlife policies.

Introduction

Travel and tourism is estimated to contribute 9.2% and 2.6% directly to the global and African gross domestic product (GDP) respectively (WTTC, 2012, WEF, 2011). Africa is one of the fastest rising tourist destinations, with annual visitor numbers projected to increase from the current 31 million to 39 million by 2014 (EMI, 2010). Africa's relatively small global market share (<4%) and resource endowments have generated an upsurge in policy attention in the tourism sector (EMI, 2010, Christie and Crompton, 2001). In Kenya, the government has identified tourism as a key pillar of "Vision 2030", the national development blueprint whose short-term target was to quadruple tourism's contribution to the GDP and increase the number of international visitors from 1.8 million in 2006 to 3 million by 2012 (Government of Kenya, 2008a).

Worldwide, one of the fastest growing tourism sub-sector is nature-based tourism (NBT) which, across southern Africa, is estimated to provide the same levels of revenue as farming, forestry and fisheries combined (Biggs et al., 2004). This trend highlights the importance of NBT as an ecosystem service (Millennium Ecosystem Assessment, 2005b) and also the growing demand by society for cultural ecosystem services generated in tropical savannah landscapes (Greiner et al., 2009). A significant component of NBT comprises wildlife tourism defined as the tourism based on encounters with non-domesticated animals, which can occur in either the animal's natural environment, or in captivity. It includes non-consumptive utilization such as wildlife viewing, photography and feeding, as well as those that involve killing or capturing of animals such as hunting in terrestrial environments and recreational fishing in aquatic environments (Higginbottom, 2004, p. 2).

Wildlife tourism is the mainstay of the national tourism industry in many of the countries in East and Southern Africa, including, in alphabetical order, Botswana, Kenya, Mozambique, Namibia South Africa, Tanzania and Zimbabwe which are some of the major international destinations (Valentine and Birtles, 2004) where wildlife tourism is based on exceptional diversity and concentrations of large mammal species scattered across savannah landscapes (Millennium Ecosystem Assessment, 2005a). In Kenya, wildlife tourism is a core product line of the tourism industry (World Bank, 2011), attracting 80% of the country's tourists and accounting for an estimated 70% of the gross tourism earnings (Ikiara and Okech, 2002).

Although highly dependent on protected areas, a recent shift in wildlife tourism is the expansion of the industry into private land with an increasing number of private landowners (communities, individuals and corporate institutions) managing or leasing their land to conservation agencies and corporate tourism enterprises for wildlife viewing and recreation (World Bank, 2011, Carter et al., 2008, OECD, 2003). This phenomenon is rapidly increasing in southern Kenya in the pastoral areas occupied by the Maasai ethnic communities leading to over-supply of tourism services (Chapter 3).

Some of the wildlife tourism arrangements between investors, conservation NGOs and government agencies on one side, and the pastoral landowners on the other involve direct payment for biodiversity schemes (Ferraro and Kiss, 2002, Jenkins et al., 2004). In these PES schemes, tourism is commonly classified as a "cultural ecosystem service"; the non-material benefits obtained from ecosystems (TEEB, 2009, Millennium Ecosystem Assessment, 2005a). However, because ecosystem service markets for wildlife tourism are a sub-set of nature based tourism (NBT), they are thus also directly linked and are highly synergistic to ecosystem service markets for landscape beauty and recreation (Lindsey et al., 2007, FAO, 2007, Bishop et al., 2008, Swallow et al., 2009).

Surprisingly, the literature on the linkages between PES and tourism remains thin despite rapid proliferation of PES schemes in which funds generated from tourism are (or could be) used to pay for ecosystem services (Ritsma et al., 2010, Huberman, 2009, de Groot, 2011, Pagiola, 2008, Clements et al., 2009). In PES schemes involving tourism, landscape beauty and recreation, low-income land managers are compensated for their stewardship of landscapes or wildlife that have scenic or recreational values to tourists, hunters, or fishers. These schemes are especially common in and around protected areas where local communities may receive a portion of visitor fees in exchange for helping to protect or refraining from harming wildlife or scenic values. Alternatively, they may receive payments directly from tourist operators for maintaining certain land uses or protecting or enhancing geographic features or charismatic species that appeal to tourists (Milder et al., 2010, Jenkins et al., 2004, Skonhoft and Solstad, 1998).

There are several examples of PES linked to tourism in general and wildlife tourism in particular. An example of the former is found in Bhutan, where tour operators are paying local communities for tourism services that put a halt to encroachment on forest areas (Ritsma et al., 2010). An example of the latter occurs in Cambodia, where an ecotourism PES scheme is operational to conserve the globally threatened large water birds in Kulen Promtep Wildlife Sanctuary. Here the provision of tourism revenue to the local community is subject to them ceasing hunting of key species and abiding by an agreed land use plan. In return, all visitors to the sanctuary pay US$ 30 per person if they see all the key species or US$ 15 per person if they observe only a sub-set of the key species (Clements et al., 2009).

A review of literature shows that in the ASAL rangelands in Africa, only a handful of such PES schemes involving wildlife tourism exists (Naidoo et al., 2011, Nelson et al., 2010, Frost and Bond, 2008). This could be due to the fact that PES has so far not been widely implemented in rangelands in general (Duttilly-Diane et al., 2007, Greiner et al., 2009, Goldstein et al., 2011). The common examples of PES and "PES-like" schemes (Engel et al., 2008) in African rangelands include: (1) the CAMPFIRE programme in Zimbabwe where safari operators buy the rights to bring eco-tourists and sport hunters to their concession areas to hunt a set quota of animals, or track, observe or photograph animals (Frost and Bond, 2008); (2) the Community Wildlife Conservancies in Namibia, in which private licensed hunting and tourism companies compete to acquire the rights to photographic safaris and trophy hunting and various plant products on communal lands (Naidoo et al., 2011); and (3) the Terrat PES scheme in northern Tanzania, in which a consortium of five tourism operators pay pastoral communities to enforce voluntary restrictions on agricultural cultivation and permanent settlement within a critical wildlife dispersal area outside Tarangire National Park. The tourism operators here pay for land in which they have no direct commercial interest, but which is of indirect value to their business operation (Nelson et al., 2010, Sachedina and Nelson, 2012).

Wildlife based tourism and pastoral poverty: a review of policy in Kenya

Poverty reduction as a multi-sectoral policy goal

Tourism is a cross-sectoral industry which is underpinned by numerous policies, legislations and regulations. In Kenya for example, a recent study found that the tourism sector is under regulation of 44 different legislative instruments (World Bank, 2011). As concerns the wildlife tourism sub-sector, the key policies involve those that regulate wildlife conservation, tourism, land use and overall development planning (Sindiga, 1995, Republic of Kenya, 2006, Homewood, 2009, Ikiara, 2001). While each of these sectors has different specific policy priorities, an objective across all the policies is that of poverty reduction, which is underpinned by the overall national development blueprints. The critical role and potential of tourism in general, and specifically wildlife tourism in economic growth and poverty reduction in Kenya is a key theme in all the post-2000 government development plans, notably the Poverty Reduction Strategy Paper (International Monetary Fund, 2005), the Economic Recovery Strategy for Wealth and Employment Creation (Government of Kenya, 2003) and the Kenya Vision 2030 (Government of Kenya, 2008a).

The policies in both the tourism and wildlife sectors have since independence included poverty reduction as a key objective (Ikiara, 2001, Homewood, 2009). The current Kenyan tourism policy for example, explicitly states that tourism shall contribute significantly towards poverty alleviation, and in particular will be a major vehicle for job creation, poverty reduction and wealth creation. It further states that the government shall encourage the involvement of local communities in managing wildlife so as to ensure that they receive a significant share of the benefits from wildlife based tourism (Republic of Kenya, 2006). Similarly, recent draft versions of the wildlife policy have all expressed the need for equitable sharing of wildlife benefits, and pay due regard to pastoral communities living within wildlife areas (Republic of Kenya, 2007, Republic of Kenya, 2011a). However, a 1977 ban on hunting of all wildlife other than game bird species currently constraints pastoral landowner's wildlife revenue options to tourism game viewing and photography only (Norton-Griffiths, 1998, Norton-Griffiths, 1996).

Also, with regard to land in the pastoral regions, the Government of Kenya has pursued a policy of privatization and land sub-division. This process was accelerated first through the enactment of the Land (Group Representative) Act of 1968 which established Group Ranches on pastoral land previously held in trust by the government (Lenaola et al., 1996). The process was followed by the sub-division of communally held land into small parcels each with an individual title deed (Lenaola et al., 1996, Galaty, 1994a).

A key policy objective underlying land privatization and sub-division in rangelands was to open up opportunities for lines of credit to attract economic investments in land management (Galaty, 1992). Contrary to this expectation, the outcome has largely contributed to a deepening poverty among a large number of pastoral households in wildlife rich and high tourism zones (Galaty, 1999, Mwangi, 2007b, Homewood, 2009). As a result, one of the stated policy objective of the recently formulated national land policy is to secure rights over land by facilitating access to land administration by the poor to enable the sector contribute more effectively to poverty reduction (Ministry of Lands, 2007).

Impact of wildlife tourism on poverty reduction and livelihoods

Community conservation (Western and Wright, 1994), including wildlife tourism is a popular strategy for wildlife conservation (Ashley and Roe, 1998, Kiss, 2004a) and poverty reduction (Manyara and Jones, 2007, Ashley and Elliot, 2003). The local communities can benefit from wildlife and tourism in several ways, including provision of alternative sources of income, products, and social benefits. Apart from operating wildlife tourism enterprises, a range of mechanisms exist through which the local communities can access these benefits. These include direct income from leasing land to tourism operators, employment in tourism enterprises, charging access, wildlife viewing and concession fees, and in through protected area revenue sharing schemes, and (Ashley, 1995, Ashley and Roe, 1998).

There is currently little evidence of major positive welfare effects of Community-based wildlife tourism initiatives on pastoral communities in Kenyan ASAL (Homewood et al., 2009c), although in a few, isolated cases, some limited livelihood benefits have been reported (DeVeau and Marshall, 2008, Sikoyo et al., 2001, Waithaka, 2002). The majority of these initiatives have however, registered marginal or negligible poverty reduction and livelihood benefits altogether (Manyara and Jones, 2007, Kellert et al., 2000, Coupe et al., 2002, Rutten, 2004). At the same time, their effect on the reduction of declines in wildlife populations outside protected areas has been minimal (Kiss, 1990, World Resources Institute, 2007). Consequently, pastoral areas around wildlife protected areas with high tourism visitation and revenues, such as the Amboseli and Maasai Mara National Reserves in Kenya, have not only witnessed serious habitat degradation leading to massive declines in wildlife populations (Western et al., 2009, Ogutu et al., 2011) but also exhibit high poverty levels among pastoral land users (Okello et al., 2009, World Resources Institute, 2007, Homewood et al., 2009b, Ogutu, 2002).

Tourism undoubtedly contributes to economic growth at the national level, in terms of its share of GDP (DFID, 1999, ODI (Overseas Development Institute), 2006), and is the sector that offers the greatest wildlife-related growth opportunities in Africa (Ashley and Elliot, 2003). Despite this asserted potential, there is scant evidence of a substantial contribution of wildlife-based tourism to poverty reduction at the local level in Kenya, particularly in pastoral areas (DFID, 2002, Coupe et al., 2002, Kiss, 2004a, Homewood, 2009, Sindiga, 1995). Norton-Griffith and others argue that the reason that wildlife tourism has provided limited benefits to pastoral communities is because of low and uncompetitive wildlife returns resulting from a combination of policy, institutional and market failures (Norton-Griffiths and Said, 2010). In particular, market failures concerning the provision of wildlife goods and services occur because of the diversion of a major portion of wildlife generated revenues away from the producers of wildlife – pastoral landowners – to the service side of the industry (Norton-Griffiths and Said, 2010, Norton-Griffiths, 2007a, Earnshaw and Emerton, 2000). This situation leaves pastoral landowners with the highest wildlife costs and risks but with no commensurate level of benefits (Bojo, 1996, Muller and Albers, 2004).

The Maasai Mara Ecosystem

Importance for wildlife and tourism

The Maasai Mara Ecosystem (MME) covers an area of 6,500 km^2, which includes the Maasai Mara National Reserve (MMNR; 1,530 km^2) and the private and communal lands adjoining the Reserve to the North and East (Lamprey and Reid, 2004). The MME is a critical part of Kenya's wildlife tourist industry (Akama, 2002, World Bank, 2011). In the Kenya *Vision 2030* medium term plan for 2008-2012, the MMNR is listed as one of the tourism flagship projects that is to be developed to provide a premium, high end tourism experience in a top wildlife destination (Government of Kenya, 2008a, NCC & TCC, 2009).

Two factors make the MME critically important for wildlife tourism. First, the MME supports the highest density of both wild and domestic herbivores in Kenya, and about 25% of the total wildlife population in Kenya is currently found in the MMNR (Western et al., 2009). Second, because of its relatively higher rainfall, grassland productivity, and permanent water sources, the MME serves as a dry-season refuge for the Serengeti's migrant wildebeest *Connochetes taurunus* and zebra *Equus burchelli* (Fryxell, 1995). The annual northwards migration of these wildebeest is a major tourist attraction in the area (Honey, 2009). Between 2001 and 2005, the MMNR alone accounted for about 13% of all international visitors to Kenya in (World Resources Institute, 2007). The majority of these visitors were accommodated in the lodges and facilities located on private land outside the Reserve (NCC & TCC, 2009, Earnshaw and Emerton, 2000).

Currently the sustainability of wildlife and tourism in the MME is threatened by major land use changes, including expansion of both large-scale, mechanized and small-scale agriculture (Norton-Griffiths, 1996, Homewood et al., 2001, Serneels et al., 2001), human population growth , spread of settlements (Lamprey and Reid, 2004), and privatization and sub-division of rangelands from large parcels under collective to small parcels under individual and corporate tenure. The process of land sub-division is driven by the desire of landholders to secure legal title and user rights to land (Galaty, 1994a). Currently, the MME is listed in the Fourth Schedule of the 2011 Draft Wildlife Bill, as a "*Critically Endangered Ecosystem*". This category is reserved for ecosystems and habitats that are considered to face the highest levels of threats (Republic of Kenya, 2011b).

Poverty and inequitable distribution of tourism benefits

Although MMNR ranks among the highest-earning protected areas in Kenya, generating US$ 15-25 million per annum (Norton-Griffiths, 1998), the majority of the landholders living around the Reserve have failed to benefit to any great extent from the thriving wildlife tourism due to skewed distribution of wildlife revenues (Earnshaw and Emerton, 2000). Tourism revenue mainly accrues to the service providers, including tour operators and camp owners, and the small proportion that accrues to the local landholders was differentially distributed among households and wealth categories; the top wealth quartile consistently took around 60-70% while the bottom quartile received around 5% of all wildlife income in 1998-2004, whether from wildlife associations and campsites, or from associated wages and business revenue (Thompson et al., 2009).

Government survey reports show that an estimated 69,000 out of a total population of 108,000 people living within 25 km of the MMNR have incomes below the Kenyan rural poverty line. Further, the prevalence of poverty rate is 63% and the poverty gap is 15-20%. The poverty rate (also known as "headcount ratio") in this case represents the percentage of the total population living below the 1999 Kenyan rural poverty line of US$ 0.59 per capita per day. The poverty gap (also known as the depth of poverty) in this case represents the average expenditure shortfalls for the poor relative to the rural poverty line (World Resources Institute, 2007, Central Bureau of Statistics, 2005). Poverty is thus widespread among pastoralists who also bear the largest cost of wildlife presence outside the MMNR, in terms of livestock predation, human deaths and injury and competition as well as restricted access to grazing and water (Norton-Griffiths et al., 2008, Omondi, 1994, Walpole et al., 2003).

Wildlife Conservancies in the Maasai Mara Ecosystem

The participation of pastoral Maasai in wildlife tourism on lands outside the Maasai Mara National Reserve has evolved considerably since early 1960s alongside demographic changes, and changes in land tenure and land use. The Mara area is one of the few places where community conservation and ecotourism was pioneered soon after Kenya attained independence from the British in 1963 (Talbot and Olindo, 1990).

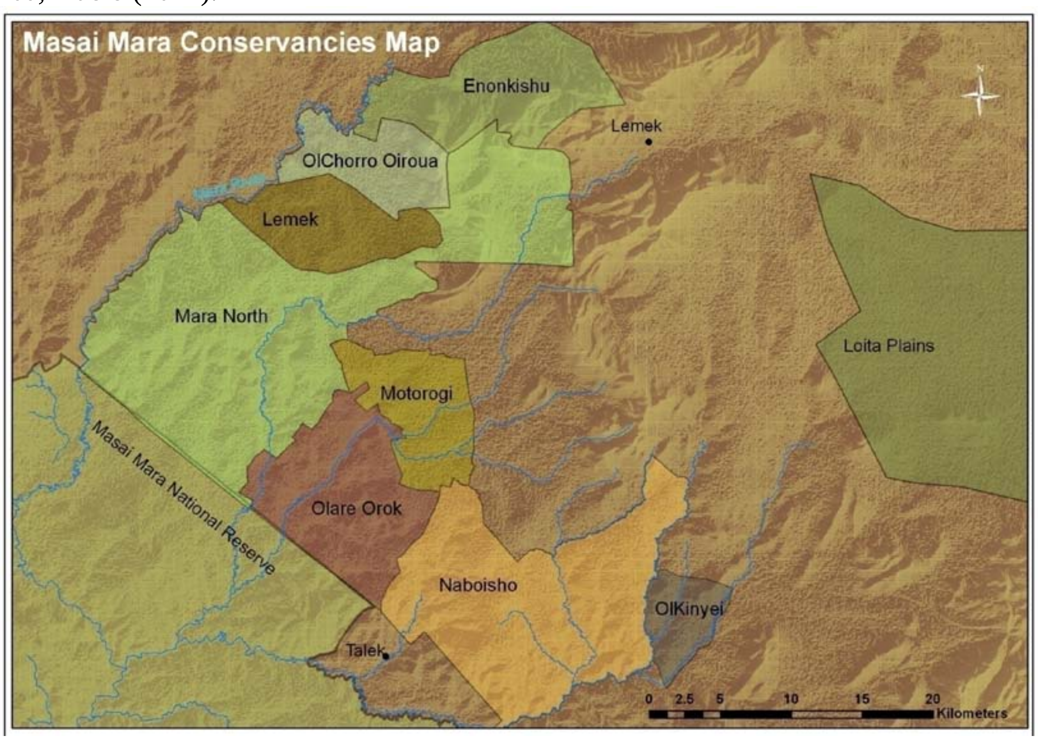
In the first decade, a Revenue Sharing (RS) scheme was established by the Narok County Council (NCC) which was granted management rights over MMNR on behalf of the local Maasai (Honey, 2009). The implementation of the RS has been controversial, mired in corruption, lack of transparency, and conflicts over how and to whom the benefits should go to, leading to its poor performance. The money from the RS scheme was invested mostly in communal projects across the entire Narok district, with little direct benefits to communities on the edge of the Reserve (Honey, 2009).

Aside from the RS scheme, tourism revenue is also generated from development and tourism user fees such as visitor entry fees, camping, vehicle use, lodge concession fees, and trophy collection fees for the lease of wildlife hunting blocks before the 1977 ban on sport hunting. Initially, these revenues were also paid to the NCC until a 1994 High court ruling declared it illegal for the NCC to tourism fees on private land, and allowed private land owners in Ol Choro Oiorua Wildlife Association to collect tourism revenue on their land (Honey, 2009, Lamprey and Reid, 2004). This decision paved way for the more than eight Group Ranches and later different Wildlife Association (WA) to collect tourism fees on the group ranch land, but again, the majority of pastoral landowners received little or no wildlife tourism benefits because the revenues were not transparently shared (Thompson et al., 2009).

Thus following the sub-division of Group Ranches, and the allocation individual titles to former Group Ranch members, commercial tourism enterprises and land owners created new institutional arrangements in the form of Conservancies to support the continuation of wildlife tourism under a privatized and individuated land tenure (Sorlie, 2008).

In the existing Conservancies in MME, Maasai landowners consolidate their individual parcels, then broker land lease agreements with a coalition of commercial tourism operators (Aboud et al., 2012a). Through the Conservancies, pastoral landowners have amalgamated adjacent plots in order to create viable game viewing areas to the north of the Maasai Mara National Reserve (Figure 6.1).

Figure 6.1 The map of the Conservancies in the Maasai Mara Ecosystem in 2010. The Olare Orok Conservancy is situated immediately adjacent to the northern boundary of the Maasai Mara National Reserve.
Source; Kaelo (2012).



Starting with only two Conservancies (Olare Orok Conservancy and Ol Kinyei Conservancy) with a combined land area of 14,576 ha in 2006, there were eight Conservancies covering an area close to 100,000 ha by 2010 (Table 6.1).

Table 6.1 Conservancies in the Maasai Mara Ecosystem (MME) and the rates (2010) for landholder's payment in the Conservancies.
Source: Lead author's compilation

Conservancy	**Year of Establishment**	**Area (Ha)**	**PES rate (KES /ha/year)**
Enoonkishu Conservancy (EC)	Not established	6,566	Not yet determined
Lemek Conservancy (LC)	Not established	6,860	bed-nights based
Ol Chorro Oiorua	Not established	6,879	bed-nights based
Olare Orok Conservancy (OOC)	2006	9,720	3000 ($43)
Ol Kinyei Conservancy (OKC)	2006	4,856	1300 ($19)
Motorogi Conservancy (MC)	2007	5,466	2500 ($36)
Mara North Conservancy (MNC)	2009	30,955	2500 ($36)
Naboisho Conservancy (NC)	2010	20,946	2000 ($29)
Total		**92,248**	

There is considerable variation in the revenue collection, management and benefit sharing arrangements. In Conservancies with operational PES schemes, pastoral landowners are not paid based on tourism a bed-night fee, which is prone to fluctuations, but based on land leases which provides guaranteed payments and is not tied to the number of tourists hence revenue generated from the bed-night and other fees. There are four models of Conservancy management that have emerged in the MME. The first model involves management by member landowners through an individual or committee. The second model involves the employment of an external Manager. The third model involves contracting a professional management company, and the fourth model involves joint management between landowner members and the tourist partners.

Despite the phenomenal growth of conservancies in the MME over a relatively short period, the effects on household poverty and livelihoods of this new model of PES for wildlife tourism remains to be assessed. Such an assessment would help provide an understanding of why Maasai landowners in the newly subdivided lands prefer to invest in wildlife tourism rather than continue with pastoral livestock production or other forms of land use on their newly sub-divided lands. Are they doing this because of persuasion by outside forces interested in using their land for conservation and tourism, or is it a case of the “great experiment” (Hopcraft, 2000) to keep lions instead of cattle?.

This paper evaluates the PES scheme operational in Olare Orok Conservancy (OOC) where Maasai landowners are paid by tourist operators to voluntarily relocate their settlements and exclude livestock grazing. Accordingly, this analysis is restricted to a specific situation where private tourism industry pays households directly for biodiversity conservation, landscape beauty and recreation (Milne and Niesten, 2009). We address the following four questions:

1) How has land tenure evolved in the Maasai Mara Ecosystem?
2) What is the nature and design of the PES scheme in the Olare Orok Conservancy?
3) What is the level of poverty and wealth inequality among households in the study area?
4) What are the effects of the PES on household poverty, inequality, income and expenditure?

Olare Orok Conservancy: The background

The Olare Orok Conservancy is one of the eight Conservancies in the MME. It is bordered on the eastern boundary by the Mara North Conservancy (MNC), to the South-west by Maasai Mara National Reserve (MMNR), to the South-east by the Naboisho Conservancy and to the North-east by Motorogi Conservancy, which is jointly managed together with the OOC (Figure 6.2 above). The area in which the Conservancy is land is located in is characterized by open grassland and acacia woodland savannah. The area contains resident wildlife and also serves as dry season dispersal areas for wildlife herds from the Maasai Mara National Reserve. The OOC encompasses lands straddled by the Olare orok and Ntiakitiak Rivers and formerly used as a dry season grazing area for the Maasai livestock in Talek, Olkurroto and Nkorbob.

The OOC was started in May 2006 in the Olare Orok area after the sub-division of land in the former Koyake Group Ranch into plots of an average size of about 150 acres (60 hectares) allocated to 1,020 members (Lamprey and Reid, 2004). It started originally as a partnership between a group of 154 new pastoral landowners and four high-end tourist operators/partners. It was established on a business model where the tourism partners lease the land and guarantee landowners a fixed payment regardless of the number of tourists visiting the camps in the conservancy. In 2007, the Motorogi was established to the east of the OOC and arrangements were made to have a joint management of these two conservancies although they remain distinct and separate in terms of their membership and governance composition.

There is very little documentation so far, about how the OOC landowners ended up with contiguous parcels of land in one of the most prime tourism areas of the MME. Previous studies on the OOC conducted by two Masters students notes that although the principle of land sub-division in the Koyake Group Ranch was for all members to get an equal share of land close to where they had lived, this was not followed to the later. There were cases where some elites from the community who had access to the group ranch register and a map of the location of plots manipulated the land sub-division process to their advantage. These elites allocated themselves the most prime land, located close to tourist facilities, permanent access to water and better pastures (Courtney, 2009, Sorlie, 2008). This thesis did not include an assessment of the politics and dynamics of the land sub-division process in the OOC, and this remains a critical area of future research, not just on the OOC and Koyake Group ranch but across all the Group Ranches in the Narok County that have already conducted, or are in the process of conducting land sub-division.

In the current OOC arrangement, the tourism partners provide funding for running the Conservancy operations and support the landholders company through the *Ol Purkel Ltd* a not-for-profit professional management company, jointly owned by the four tourism partners and the landowners through the landholders company. Pastoral land owners have to adhere to two conditions when they enroll land in the Conservancy.

First, the landowners are required to relocate their settlements outside the conservancy area. Manu of the land owners have moved and settled in the nearby trading centers such as Talek, Sekenani and Aitong (Philip Osano personal observation). Others have relocated their settlements to yet to be sub-divided lands, or on land owned by relatives and clan members. Sorlie (2008) documented that the decision to remove settlements was not only a controversial one among land owners, but was also costly beyond the reach of the majority of pastoral families, who had to rely on financial support from the elites:

> "Some landowners lived on other peoples land and could easily be removed. Other landowners held pieces of land in other areas of Koyiaki and could move their bomas to these parcels. The process was greatly simplified by the effort of local elites who assisted landowners with the costs of moving their buildings and belongings. After three months all the bomas were removed from the area"
> (Sorlie, 2008, p. 60)

Second, the landowners are also not allowed to graze inside the Conservancy land without the permission of the OOC management. The exception was during droughts when the Conservancy management allowed group grazing under a tightly controlled grazing schedule enforced by *Ol Purkel Ltd*. The limitation on grazing has reduced the area available for livestock pasture, and thus increased livestock pressure on land (ILRI unpublished data on livestock population changes in the MME). Furthermore, the grazing regulation is also a controversial one because it affects not just the Conservancy members alone but also non-members as well. Courtney (2009; 34) notes that since some of the OOC members live and graze for free on the land that non-members are currently waiting to be allocated, non-members should therefore not be excluded from grazing in the OOC during the dry season.

The payment to landowners in the OOC has been structured in phases. In the initial phase of establishment in 2006, contracts of one year were provided to landowners, with PES rates[5] of KES 2500/ha/yr (US$ 33/ha/yr). This has since been gradually revised in the implementation phase with rates of KES 3000/ha/yr in 2009 (US$

[5] Exchange rate of KES 1=US$ 0.013 in 2006; 0.012 in 2009 and 0.010 in 2011 respectively (Source: www.oanda.com)

36/ha/yr) and KES 3750/ha/yr in 2011 (US$ 41/ha/yr). The revisions of rates are to account for inflation, changes in contract options (1, 5 and 15 years respectively) based on the negotiations between landowners and the tourism partners. The land rental fee is paid directly to the households that unless otherwise, are required to open bank accounts where the money is deposited by *Ol Purkel Ltd*.

Sorlie (2008) analyzed the process of OOC establishment and concluded that the OOC was created because landowners and tourism entrepreneurs saw an opportunity to profit, because they had the necessary resources to establish the conservancy and because the institutional setting provided an opportunity for change (Sorlie, 2008: 88-89). This conclusion, viewed from an economic perspective, suggests that the OOC created the possibility for a “win-win” outcome for both the landowners and tourism operators.

Tourism expected to reap profits by offering their clients exclusive safari experiences in an exclusive wildlife viewing area outside the overcrowded MMNR. The pastoral landowners expected to derive a higher land rent from collectively leasing out their land to a wildlife conservancy than by using it for agricultural production or livestock grazing. Furthermore, the policy changes in land tenure and devolution of wildlife management was a critical factor enabling the establishment of OOC. In particular, the subdivision of group ranches in the Mara empowered ordinary landowners and gave them greater control over the use of their land forcing local elites and tourism operators to consider the interests of ordinary landowners in the establishment of the OOC (Sorlie 2008).

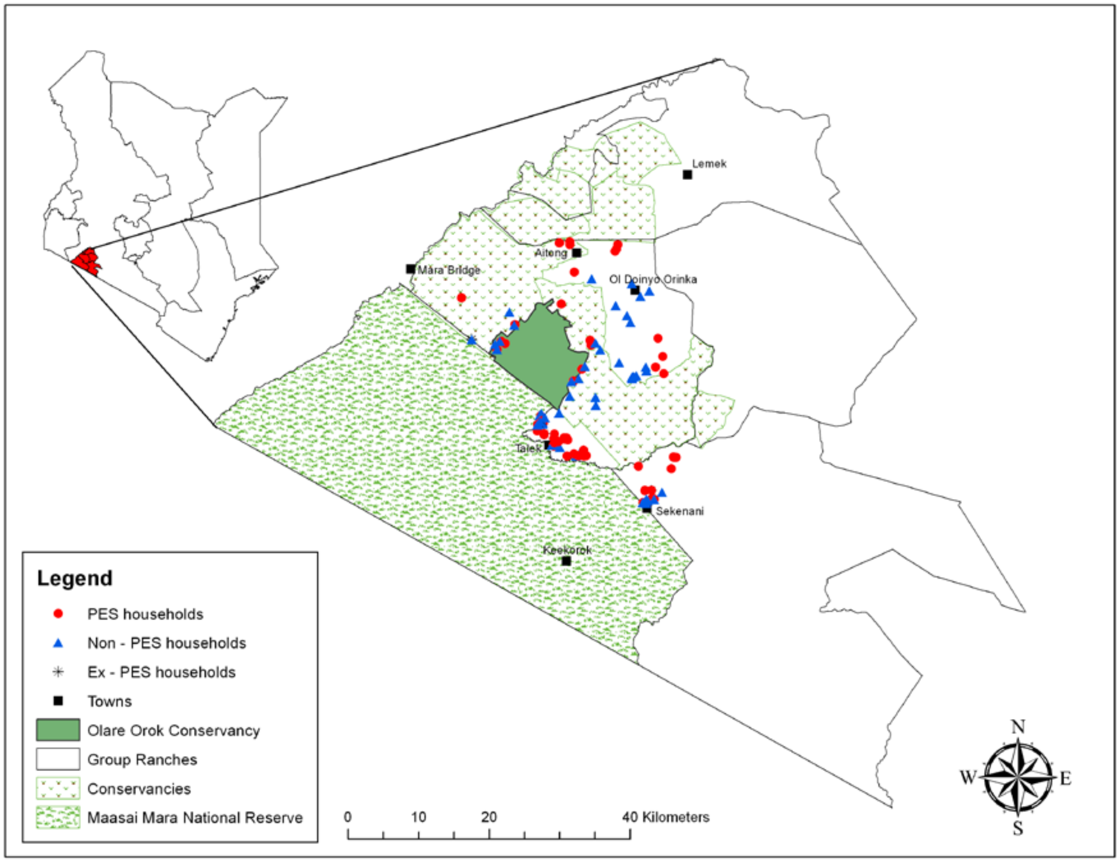
Methods

The data and data analysis

The primary data used in this study was obtained from the following sources: i) household survey carried out from January to April 2010. The survey included a total of 131 households selected by random sampling (Figure 6.2). The sample comprised 73 OOC households and 58 households with land outside Olare orok area, hence not enrolled in the OOC. The survey elicited detailed information on household land ownership and land use in and outside OOC, and socio-economic data, including demographic characteristics, cash income in 2008-2009 from livestock, PES in form of wildlife rents, cropping, trade and remittances, and the location of households based on the Global Positioning System (GPS) recording; ii) Semi-structured and informal interviews carried out by the lead author with landowners, officials of *Ol Purkel Ltd*, community members and key informants; iii) Focus group discussions with representatives of the tourist partners in the OOC, Motorogi and Naboisho Conservancies carried out in August 2011; iv) A national workshop on Payments for Ecosystem Services and Conservancies held in February 2012 that brought together pastoral landowners, conservancy managers, policy makers in governments, conservation NGOs and researchers. The workshop proceedings is published as an report of the International Livestock Research Institute (Bosire et al., 2012).

The secondary data used were gathered from the following sources: i) A literature review of publications in the peer reviewed journals, workshop reports and other grey literature; ii) Institutional databases provided by *Ol Purkel Ltd*, the International Livestock Research Institute (ILRI), and the Department of Resource Surveys & Remote Sensing (DRSRS). The *Ol Purkel* Ltd provided information on OOC enrolment, land ownership, livestock incursions within the OOC and copies of the land lease agreements with landholders. The ILRI database contained geo-spatial data on land cover and land use, water sources (rivers, dams), urban areas, social amenities (health centers, schools, markets), and infrastructure (roads). The DRSRS provided data on livestock (goats, sheep and cattle) numbers in the MME for the period 1977-2011; iii) The Normalized Difference Vegetation Index (NDVI) data at a spatial resolution of 8 km for the MME was obtained from the National Oceanic and Atmospheric Administration (NOAA) Advanced Very High Resolution Radiometer (AVHRR). NDVI measures the amount and vigor of vegetation at the land surface.

Figure 6.2 Map of Maasai Mara Ecosystem (MME) in south-western Kenya showing the location of the surveyed households, of the Olare Orok Conservancy, the Maasai Mara National Reserve, the Group Ranches and other Conservancies
Source; Lead author's survey.
PES Households: Households enrolled in the Olare Orok Conservancy (OOC); **Non-PES Households:** Households not enrolled in OOC; **Ex-PES Households:** Households that dropped from OOC.



We analyzed the survey data in MS-Excel, SPSS and SAS softwares, and conducted geo-spatial analysis in ArcGIS 9.3 (ESRI, Redlands, California). We excluded from our quantitative analysis 13 households in total; six households that reported having dropped out of the OOC and another seven that were not enrolled in OOC but were participating in the Mara North Conservancy Wildlife PES scheme hence could not qualify as control households. The resulting number of household considered in the quantitative analysis was 118 in total, but the perspectives of all the 131 households interviewed were considered in the qualitative analysis.

Land tenure and changing property rights

We mapped land tenure changes from before independence in 1963 to 2012 to visually represent the transition from trust lands to private land tenure, including group ranches, sub-divided plots, and reconsolidated parcels in the newly established Conservancies in the MME.

The design and implementation of the OOC PES scheme

We evaluated the OOC based on four elements of an 'idealized' PES scheme: voluntary transaction; actors; ecosystem services; and conditionality (Wunder, 2005, Wunder, 2007). In assessing the conditionality for restriction on settlements, we built upon and extended the temporal coverage of the of permanent and temporary settlement distribution maps for MME developed to cover changes from for the period 1959- 1999 (Lamprey and Reid, 2004) to include the 1999-2012 period.

Next, using the livestock aerial census data from the Kenyan Department of Resource Surveys and remote Sensing (DRSRS), we calculated the monthly changes in the population density of sheep and goats (shoat) across the area covered by all the eight conservancies in the MME (82,000ha) for the period 1977-2011 to evaluate the dynamics and intensity of grazing pressure. We then generated the standardized vegetation index for the period 1982 to 2009 by taking the 12 month running average for the Normalized Differential Vegetation Index (NDVI) data and calculating the monthly NDVI anomalies for the entire time series using the z-transform $((x_i-\mu)/\text{std})$ equation, where;

x_i is the NDVI value for a given month in year i,

μ is the mean NDVI value for that month across all the years

std represents the standard deviation of the NDVI values for that month across all years

Poverty and inequality assessments

We first compared the differences between households enrolled in the OOC and those not enrolled using the standardized t-test and chi-square test. We assessed household poverty and wealth inequality using cash incomes and livestock assets and then established livelihood groups based on these two variables. We calculated income poverty by dividing the gross household income (2008 and 2009) by the household size, expressed in terms of adult equivalent (AE) to obtain the gross annual income/AE. The concept of adult equivalent (AE) is based on differences in human nutrition requirements according to age, where; <4, 5-14 and > 15 years of age are equivalent to 0.24, 0.65 and 1 AE, respectively (Kristjanson et al., 2002).

Thereafter, we converted the annual gross income/AE into monthly income and classified all households with an income equal to or below the monthly Kenyan rural poverty line of KES 1,562/AE as poor, and the remaining as non-poor. The Kenya poverty line of KES 1,562/AE per month in rural areas and KES 2,913/AE per month in urban areas is based on estimated expenditures on minimum provisions of food and non-food items.

We established livestock poverty by converting the populations of all the cattle, goats and sheep into Tropical Livestock Units (TLU) to allow for comparisons among the surveyed households and with other studies. We calculated the TLU by multiplying the total cattle numbers per household by 0.72 and the total shoat (goats and sheep) numbers by 0.17 (Radeny et al., 2007). We then calculated the per capita livestock ownership (TLU/AE), and grouped households with livestock equal to or less than 4.5

TLU/AE as livestock-poor. We chose the 4.5 TLU/AE threshold because it is considered as a level below which a pastoral household is at risk of falling into a poverty trap (Lybbert et al., 2004).

We then plotted a graph of income versus livestock holdings for all households surveyed for 2008 and 2009, distinguishing OOC from non-OOC households, in order to determine the distribution of households by wealth and poverty status as assessed by a combination of cash and livestock holdings. Using the data for 2008, we stratified households into four livelihood groups by separating households above and below the median in terms of herd size and cash income. The wealthy group (designated **W**) consists of household with above median income and livestock holdings and the poor group (designated **P**) comprised of households with below medium cash income and livestock holdings. A third middle group (designated M1 and M2) formed a middle group. M1 consisted of households with above median livestock holding but below median cash income, while M2 consisted of households with below median livestock holdings but above median cash income. The P group therefore represents the most vulnerable pastoral households that are likely to fall into a "poverty trap" with limited potential to escape (McPeak et al., 2012).

To assess household wealth inequality, we first calculated the gini-index for cash income (2009), livestock (cattle, goats and sheep) and land ownership following the method developed by Araar and Duclos (Araar and Duclos, 2009) and then tested for the differences in the gini-coefficient between OOC and non-OOC households based on standard error estimates (Russell, 2009). We also calculated and then compared the percentage coefficient of variation (CV) for the different sources of household income. High % CV scores indicate high levels of inequality and vice-versa.

Assessment of the effects of PES on household cash incomes

We assessed the effects of PES on the cash income of OOC households using two methods. The first, by estimating the magnitude of PES transfers in proportion to the poverty gap, and the second, by calculating the relative contribution of PES to the gross household income in 2008 and 2009 (Kosoy et al., 2007, Grieg-Gran et al., 2005).

Assessment of the effect of PES on household expenditure

We asked each respondent enrolled in OOC to provide an estimate of household expenditure on PES income earned in 2009 for seven bundles of goods and services (GSs), namely: food and basic needs (grains, sugar, tea, milk, cloths etc); water purchase for domestic use; human health expenses (purchase of drugs, treatment costs); educational expenses (school fees and uniforms, books and pens, etc.); veterinary expenses (drugs, vet costs, etc.); livestock purchase (cattle and shoats); and hay purchase or lease of grazing rights.

Using the expenditure estimates, we computed the household PES budget by summing the total expenses across all the seven bundles of goods and services, and the estimated household per capita (AE) expenditure for all.

Results

The evolution of land tenure policy in MME

Four major changes in land tenure have occurred affecting land tenure and land use management in the MME since 1910 (Figures 6.3a-d) (Homewood et al., 2009a). The first major change was the establishment of the Maasai Mara National Reserve (MMNR: Figure 6.3a) following the appropriation of sections of MME for protection of wildlife (Talbot and Olindo, 1990). This action effectively converted the MMNR into a formal conservation estate in which the property rights to land are vested in the Government of Kenya (Norton-Griffiths, 1996).

The second major change was the enactment, in the late 1960s, of the Group (Lands) Representatives Act (GRs, 1968) which provided a legal framework for the establishment of the Group Ranches as part of the rangeland privatization process (Figure 6.3b; (Mwangi and Ostrom, 2009b). The period from 1972 to 1980s marked the creation of GRs in the Narok District, including the MME which included eight GR surrounding the MMNR.

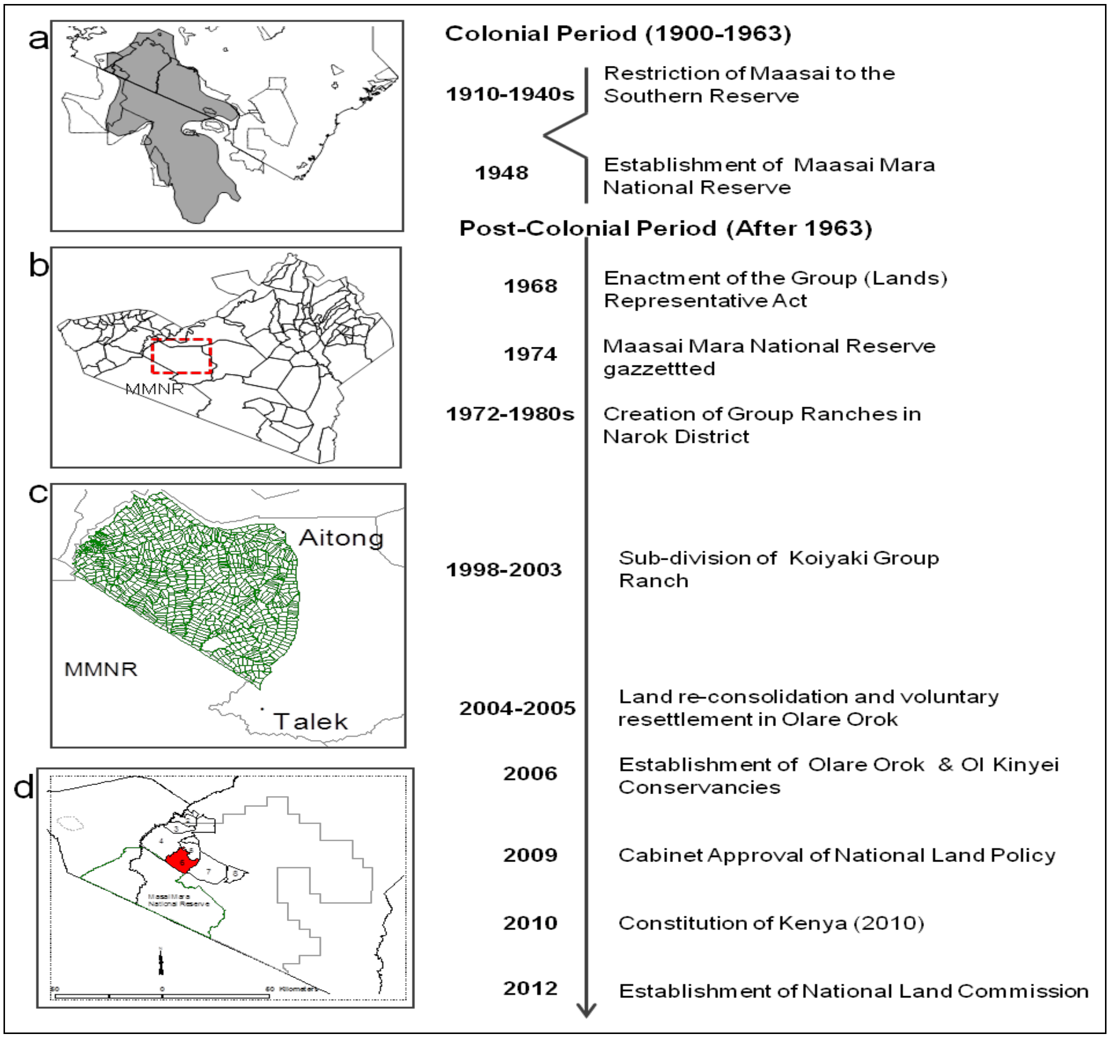
The third major change involves the sub-division of GRs from large tracts under collective property rights to small plots under individual property rights to land (Figure 6.3c). Various studies have identified three main factors driving the process of land sub-division in rangelands. One is the need for increased security of tenure to protect rangelands from in-migrants and from alienation of land by political elites, the state, and the Conservation NGOs wishing to extend the conservation estate. Two is the dilution of the value of communal resources in the face of rapid population growth. Three is because sub-division allows the economic benefits of agricultural, livestock and wildlife production to be captured directly at the household level rather than through communal institutions (Galaty, 1994a, Norton-Griffiths and Said, 2010).

The fourth major change is represented by the current consolidation of individual land holdings to establish Conservancies for wildlife tourism (Figure 6.3d). This current process of change is happening at a time of major changes in the governance and management of land in Kenya, including in the ASAL area. For the first time, A National Land Policy for Kenya was approved by the Cabinet in the *Sessional Paper No 9*, of 2009. This was followed by the enactment of a new Constitution in 2010 (Figure 5.3d) which largely re-affirmed the national land policy.

The 2010 Constitution vests all land in Kenya on the people collectively as a nation, as communities, and as individuals (Art. 61), and elevates community to a land tenure category with equal legal force and protection as public and private land. It further provides for the creation of a National Land Commission (NLC) to manage public land, and the enactment of a land statute to govern communal lands. The NLC has already been established in 2012, and the process of developing a statute by the national assembly to govern communal lands in Kenya is currently underway.

Figure 6.3 Land tenure transitions in the MME. **A**. The 21st century Maasailand (Homewood et al., 2009). **B.** The group ranches in Narok District. **C**. The Koyiaki Group Ranch sub-divided to individual plots. **D**. MME ecosystem showing Olare Orok Conservancy (6) and other post-subdivision Conservancies (1=; Enoonkishu; 2=Lemek; 3=Ol Choro Ouiroua; 4=Mara North; 5=Motorogi; 6=Olare Orok (OOC); 7=Naboisho; 8=Ol Kinyei.

Source; Compilation by the lead author



The design and implementation of the OOC PES scheme

The OOC (10,040 ha) was established in May 2006 with 157 households. In 2007, Motorogi Conservancy (4,856 ha) with 120 household was also founded to the north of OOC. Since then, both conservancies, though separate, have been jointly managed by a single company (O'Meara, 2011). Table 6.2 shows respondents' perceptions of some aspects of the OOC payments. Despite the fact that 94% of respondents indicated that they were provided with sufficient information prior to joining the OOC, 88% reported that they did not participate in the discussions in which the amount of money that is paid as land rent was determined, 84% did not know who actually set the price, and 79% did not consider the current (2010) payments to be sufficient.

Surprisingly, nearly half of all the respondents reported having been fined for violation of the conservancy rules, particularly those relating to illegal grazing.

Table 6.2 Perceptions and views of landowners regarding the PES payment features and contract arrangements (as at January 2010) in the Olare Orok Conservancy (OOC) (n=73).
Source; Lead author's survey.

Statement	**Yes (%)**	**No (%)**
Did you participate in setting the amount that is paid in the OOC (KES 2000/ha/year)?	12	88
Do you know who set the OOC payment price of KES2000/ha/year was set?	16	84
Do you consider the current OOC payment of KES 2000/ha/year to be sufficient?	21	79
Do you consider the information provided to you before joining the Olare Orok Conservancy (OOC) as sufficient?	94	4
Have you been fined for violating the OOC regulations (e.g. illegal grazing?)	49	51
Are you interested to stay in the Olare Orok Conservancy (OOC) in the next five years?	63	37

Voluntary transaction

Six of our surveyed households currently not enrolled in the OOC reported having dropped during the period between 2006 and 2010, representing an annual dropout rate of 3.9%. Many of these withdrawals were occasioned by the household's dissatisfaction with the decision to extend the OOC lease contracts from the initial 5 to 15 years.

In our survey, 67% of the OOC landholders reported having an interest to remain in the conservancy for the next five years (Table 5.3). The contracts provided from 2006 and 2009 were for two and a half years and five years respectively but these were replaced by 15-year contracts for the period 2010-2025, with a 90% acceptance rate (O'Meara, 2011). Some of the landowners that did not accept to sign for the 15-year contracts were offered the five-year contract option after a series of protracted negotiations. On the demand side, all the four founding tourism operators are still involved in the Conservancy. An additional tourism operator joined in 2009 and 2010.

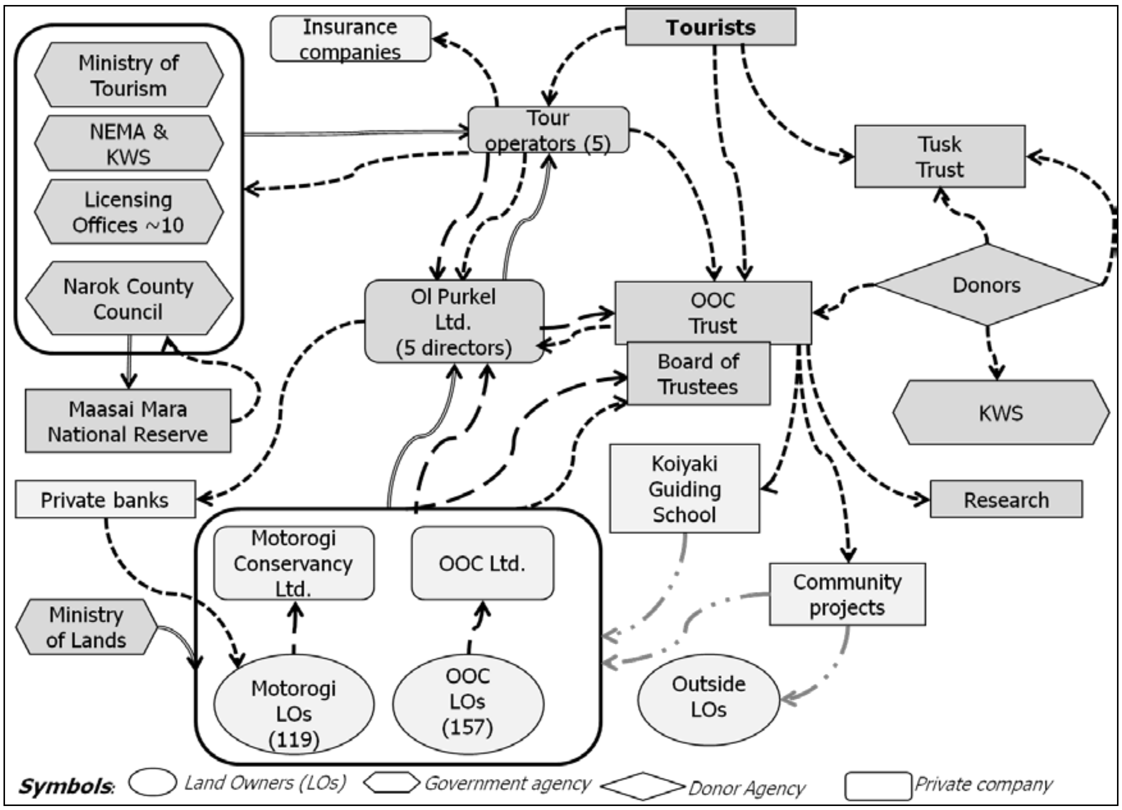
Actors and institutional arrangements

Figure 6.4 is a schematic representation of the key actors and other stakeholders in the OOC and the institutional arrangements that currently exist. It shows a complex institutional architecture with many different actors involved. The pastoral landowners with land located in the Olare Orok (and since 2007 also the Motorogi areas) are the potential ecosystem service providers in this scheme. The landowners in these two zones have both formed limited liability companies, the OOC Ltd and the Motorogi Conservancy Ltd, respectively, in which they are 100% shareholders (Figure 5.4).

The two landowners' company leases shareholders land by signing individual contracts with each landowner. The landowners companies then collectively sign a Conservancy Management Agreement (CMA) with the *Ol Purkel Ltd* which is contracted to manage the affairs of the OOC. The *Ol Purkel Ltd* is a not-for-profit company and it has an equal number of representatives of the tourism operators, the land owners companies and the OOC Trust in the Board of Directors. The *Ol Purkel Ltd* currently has 22 employees, the majority of whom are from the local community. In 2011, it had an operating budget of KES 55 million (US$ 636,000). There are thus three intermediaries in the OOC PES scheme: (1) the OOC Ltd; (2) the Motorogi Conservancy Ltd, and (3) the *Ol Purkel Ltd* (Figure 6.4).

The OOC PES scheme has five ecosystem service user or buyers, consisting mainly of the tourist camp operators. These include the Porini Lion Camp, the Kicheche Bush Camp, the Mara Plains Camp, the Olare Camp, and Virgin Camp. The operators have limited tourism in the Conservancy to a maximum of 94 beds in five mobile camps, which equates to a ratio of one game viewing vehicle for every 2,100 acres. This move is aimed at maximizing the client wilderness experience and minimizes the environmental impact of tourism (www.oocmara.com).

Figure 6.4 Institutional arrangements in the Olare Orok and the Motorogi Conservancies
Source; (Birner and Osano, 2012).
Financial transfers – dotted lines. **Representation** – dashed lines. **Regulation and/or agreements** – double continuous lines. **Non financial benefit flows** – broken and dotted lines.
*Acronyms***:** OOC = Olare Orok Conservancy. LO = Land Owners. KWS = Kenya Wildlife Service. NEMA = National Environmental Management Authority.



There are also several other actors. A key actor outside the ES buyer-intermediary-seller classification is the Olare Orok Conservancy Trust (OOCT) (Figure 6.4). The Olare Orok Conservancy Trust was set up in 2009 to administer donor funding as well as independent contributions towards community projects, together with establishing the creation and ongoing infrastructure of the Conservancy. It therefore operates outside the PES arrangements as the benefits provided to landowners through the Trust are not subject to the Conditionality of OOC PES scheme. The Trust supports additional income earning ventures, compatible with wildlife tourism and different social and development projects for the local community.

There are also several government agencies that interact with the Conservancy as shown in Figure 6.4. The major ones include the Ministry of Lands which is responsible for the issuance of title deeds to land during the sub-division process, and is also the repository of the land lease agreements. The Ministry of tourism issues licenses for all tourism facilities as well as guidelines on their operations.

The Ministry of Wildlife and Forestry through the Kenya Wildlife Service (KWS) is responsible for wildlife conservation and is in the process of developing a policy guideline for the operations of Conservancies which are lacking in Kenya (KWS, 2012). Lastly, the Narok County Council (NCC) and the Trans-Mara County Council (TCC) both of which are under the Ministry of Local Government play critical role because they are responsible for the management of the Maasai Mara National Reserve (MMNR: Figure 6.4).

Ecosystem services

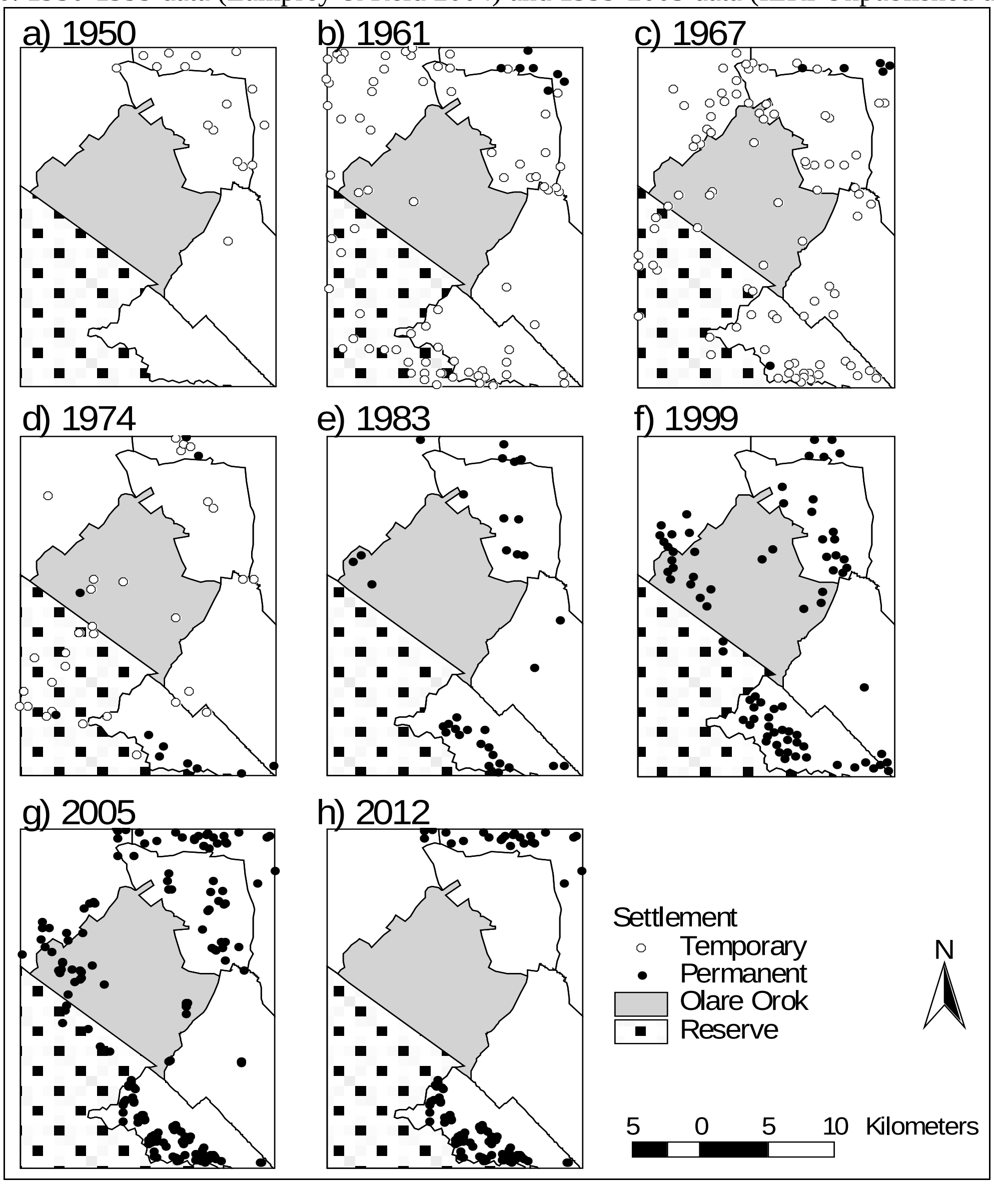
The tourist operators in the OOC lease land from the pastoral landowners for exclusive wildlife tourism uses involving eco-camp accommodation, wildlife photography, game viewing and recreation in areas of spectacular scenery. Because the operators pay the landowners for a 'premium tourism location' (O'Meara, 2011), the type of ecosystem service (ES) that is sold and bought in the OOC PES scheme can be classified as "cultural ecosystem service" of tourism (Millennium Ecosystem Assessment, 2005b). This ES depends on the "*habitat services*" (TEEB, 2010) for the presence of wildlife on land outside the protected area of Maasai Mara National Reserve (MMNR).

PES Conditionality

The OOC land lease agreement (May 1, 2010 version) developed under the Registered Lands Act (Cap 300) of the Laws of Kenya, states that the OOC land should be used "solely for wildlife conservation purposes and activities ancillary thereto including the operation of a commercial eco-tourism facility". It also includes a provision that the landowner should "not to use or permit the Premises of any part thereof to be used to graze livestock save in accordance with a grazing management plan set out by the Tenant" (Raffman Dhanji Elms Virdee Advocates, 2011). In effect, the OOC landowners have to exclude settlements and livestock from the Conservancy land.

Figure 6.5 shows the patterns of temporary and permanent settlements in the MME over the period 1950 to 2012. It shows only a few temporary and no permanent settlements between 1950 and 1967 (Fig 6.5.a, b, c). In 1974, a single permanent settlement was built (Fig 6.5.d), increasing to three in 1983 (Fig 6.5e) and to about 20 by 1999 (Fig 6.5.f). After 2005 all the settlements in the OOC were removed (Fig 6.5.g, h). Figure 6.5 also shows the progressive shift from temporary to permanent settlements from 1974 (Fig 6.5d) onwards, signaling sendetarisation from a formerly semi-nomadic pastoral lifestyle.

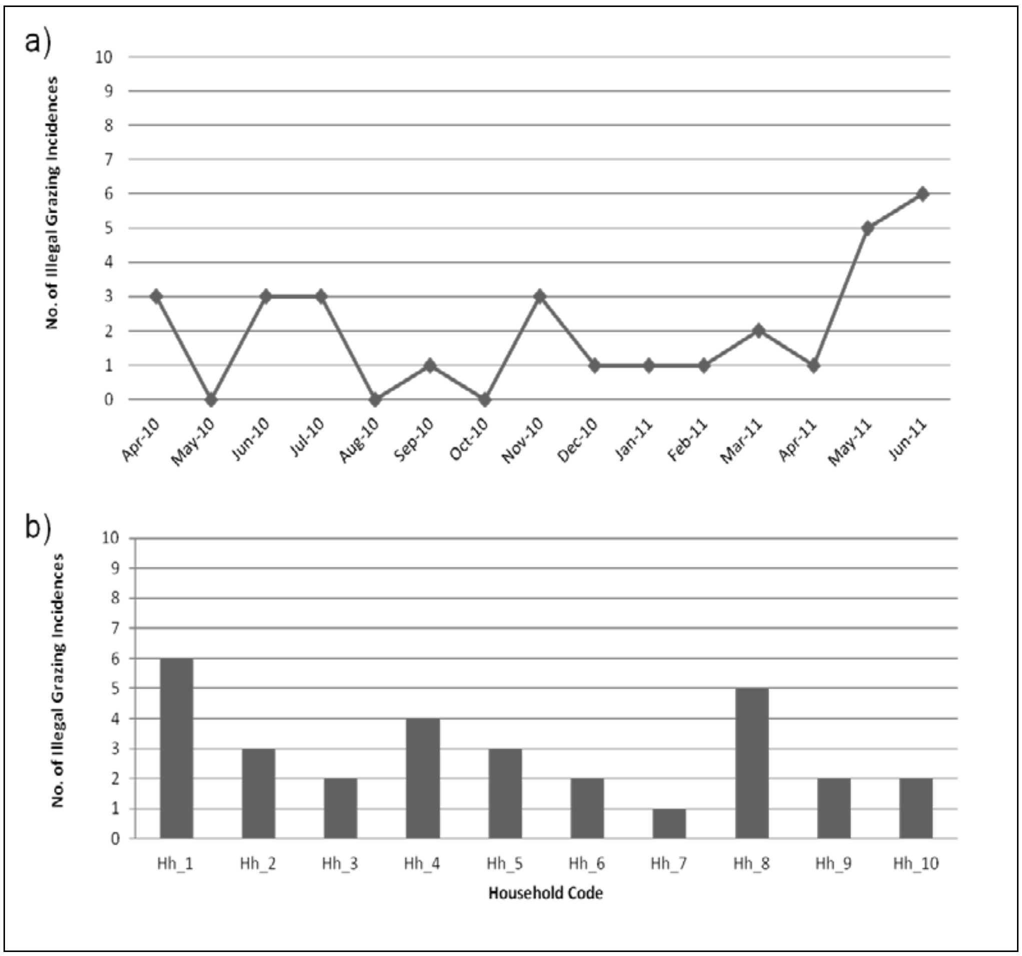
Figure 6.5 Temporary and permanent settlements in a portion of the Maasai Mara Ecosystem (MME) in the period 1950 to 2012. The dark shaded area represents the loction of the Olare Orok Conservancy (OOC) and the dotted portion is the Maasai Mara National Reserve (MMNR).
Source. 1950-1999 data (Lamprey & Reid 2004) and 1999-2005 data (ILRI Unpublished data).



There are a number of grazing violations recorded in the OOC but complete records were not available. Partial records of the number of grazing violations obtained from *Ol Purkel* Ltd are shown in Figure 6.6. It shows that a total of 10 households, including five OOC members infringed on the Conservancy in the one year period from April 2010 to June 2011. These livestock owners were collectively fined a total of KES

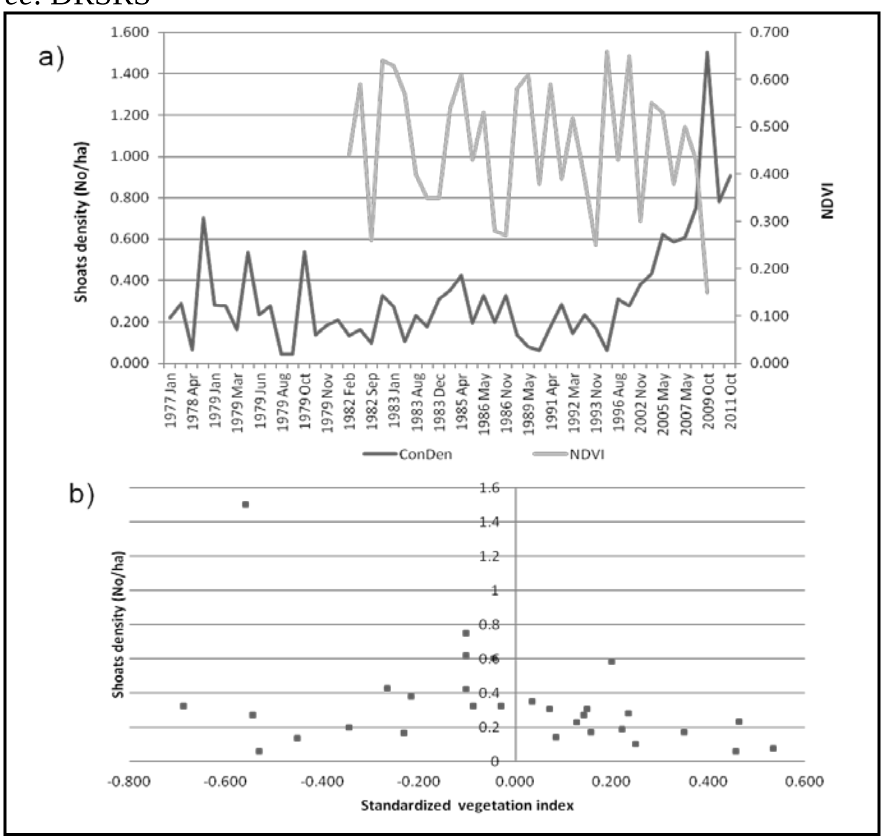
150,000 (US$ 1,660). The OOC imposes a standard fine of KES 5000 (US$ 55) per single incidence of unauthorized grazing in the Conservancy irrespective of the number of cattle involved. Through an arrangement with the *Ol Purkel Ltd*, fines imposed on Conservancy members for grazing violations are deducted from their lease payments.

Figure 6.6 Partial record of number of incidences involving unauthorized livestock grazing recorded in the Olare Orok Conservancy (OOC). **A** in the period from April 2010 to June 2011 **B** unauthorized grazing by individual households
Data source; *Ol Purkel* Ltd.



The available records show that three households (codes 1, 4 and 8) accounted for half of all the violations (Figure 6.6 b). Unauthorized grazing inside the Conservancy can result from shortage of, or poor conditions of pasture in areas outside the Conservancy, so an understanding of the conditions of pasture may illuminate this challenge for pastoral herders in the MME. As an indicator of pasture quality, the Normalized Differential Vegetation Index (NDVI) shows high habitat desiccation in the conservancy areas from January 2009 onwards, reflecting the severe drought experienced in this period (Figure 6.7). The density of sheep and goats (shoats) increased progressively from 1996 onwards and peaked in 2008, but declined steeply in 2009 and 2010 corresponding with the peak of the drought (Figure 6.7a) . Analysis of the shoat density in relation to the standardized vegetation index for the period 1982 to 2009 shows that the shoat density ranged between 0.06 to 1.5 shoats per hectare with the majority of points recording positive values along the SVI axis, representing conditions of good pasture potential (Figure 6.7b).

Figure 6.7 A The density of sheep and goats (shoats) in the Conservancy zones in the Maasai Mara Ecosystem (MME) for the period 1977-2011, and the NDVI for the corresponding months for the period 1982 to 2009. **B** The density of shoats in relation to the standardized vegetation index for the Conservancy zones in the MME.
Data source: DRSRS



Poverty and wealth inequality

Households enrolled in the OOC were significantly different from their non-OOC counterparts in terms of land size, cash income (in both 2008 and 2009), cattle (again in both 2008 and 2009) and small stocks (sheep and goat owned in 2008). Significant differences were also recorded in child dependency, and in educational levels (Table 6.3).

Table 6.3 Summary statistics (mean) for the households surveyed in the Maasai Mara Ecosystem (standard deviations in parenthesis in columns (1) and (2)).
Data source; Lead author's survey

	(1) OOC Households	**(2) Non-OOC Households**	**t-statistic (p-value)**
Continuous variables			
Cattle in 2008 (TLU)	116.88 (121.073)	79.25 (79.387)	-1.852 (0.0665)*
Sheep and goat in 2008 (TLU)	40.39 (44.380)	28.11 (26.866)	-1.674 (0.0969)*
Cattle in 2009 (TLU)	109.27 (141.653)	69.04 (73.694)	-1.7620 (0.0807)*
Sheep and goat in 2009 (TLU)	40.8 (46.206)	30.38 (33.224)	-1.316 (0.191)
Gross income in 2008 in KES (mean)	363,741.4 (230,517)	242,563.3 (189,417.6)	-2.9621 (0.0037)**
Gross income in 2009 in KES (mean)	363,082.5 (298,807.5)	184,455.6 (210,721.5).	-3.5061 (0.0006)***
Total land owned (ha)	71.08 (28.237)	46.010 (34.941)	-4.2743 (0.0000)***
Household size (adult equivalent)	17.14 (8.649)	17.64 (11.469)	0.2711 (0.7868)
Child dependency ratio (ratio)	1.81 (0.925)	1.49 (0.801)	-1.8868 (0.0617)*
Categorical variables			**Chi-square test (DF)**
Educational level (%)	18	38	5.86 (0.016)**
No. of household in sample (n)	73	45	

Notes: **P*<0.1; ***P*<0.05; ****P*<0.01.
Source: Authors' survey

Cash and livestock poverty

Overall, income poverty prevalence was higher in 2009 than in 2008 among all the surveyed households, based on both the Kenyan rural poverty line and the international poverty line (Table 6.4). Analysis of household income poverty based on the Kenyan rural poverty standards showed that 53% of all households surveyed in the MME were income-poor in 2008 and 2009, respectively. Among OOC households, 44% were income poor in 2008 and 47% in 2009. Among non-OOC households, 52% and 67% were poor in 2008 and 2009 respectively (Table 6.4). Poverty analysis based on the international measure of ‘extreme poverty’ of US$ 1 per person per day showed 50% of all MME households were extremely poor in 2008 and 66% in 2009 (Table 6.4). Among OOC households, 47% were extremely poor in 2008 and 63% in 2009. Among non-OOC households, 58% and 75% were extremely poor in 2008 and 2009 respectively (Table 6.4).

Overall, the estimates of livestock poverty among all surveyed households as well as within OOC and non-OOC households were generally low at 40% and below (Table 6.4). Similar to income poverty, overall livestock poverty among all the surveyed households was also higher in 2009 than in 2008, with estimates of 40% and 35% respectively. Among OOC households, 33% were livestock poor in 2008 but this number increased to 37% in 2009. Among the non-OOC households livestock poverty was recorded in 37% of household in 2008 and 42% in 2009 (Table 6.4).

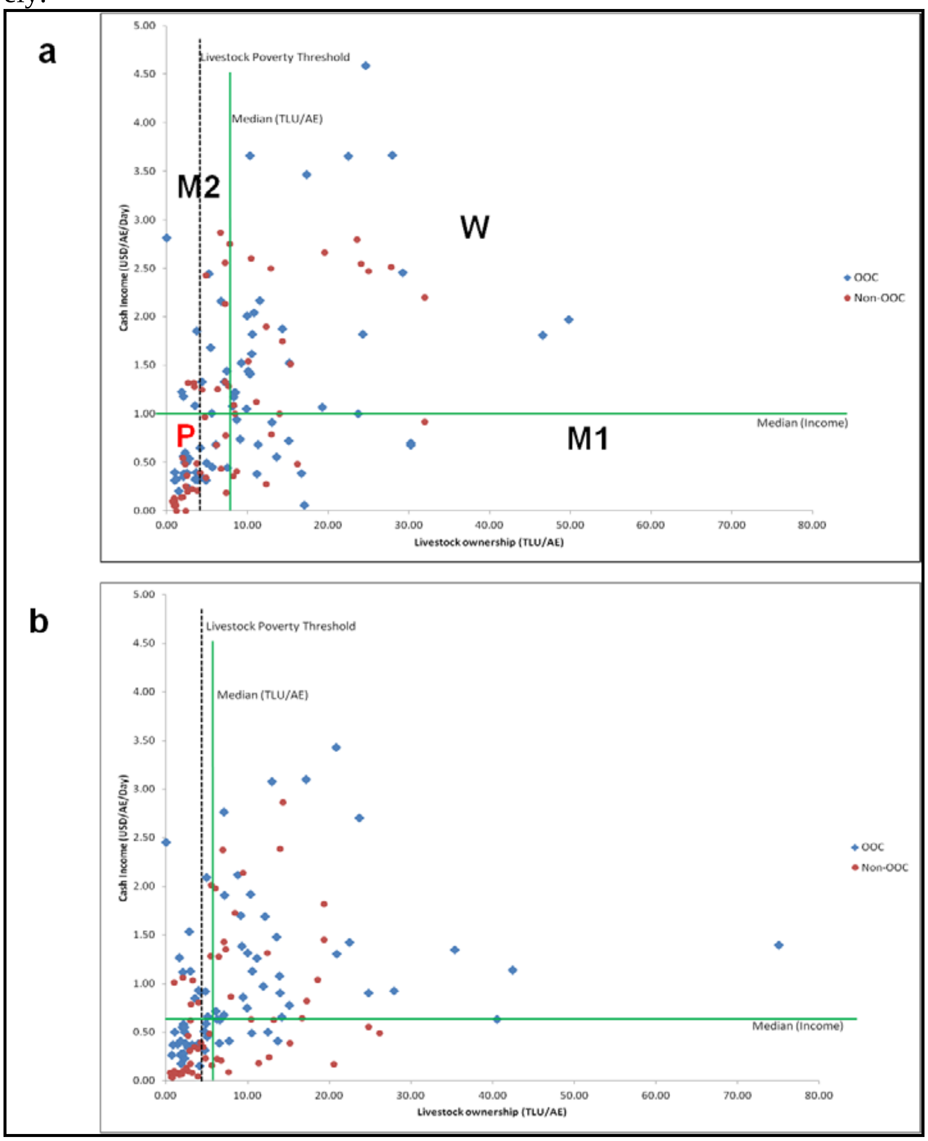
Table 6.4 Income and livestock poverty among OOC and non-OOC households surveyed in MME in 2008 and 2009
Data source; Lead authors' survey

			2008						2009					
			All HHs		OOC HHs		Non-OOC HHs[d]		All HHs		OOC HHs		Non-OOC HHs[d]	
			No	(%)	No	(%)	No	(%)	No	(%)	No	(%)	No	(%)
Income poverty	Kenya rural poverty line (KES 1,562./AE[a]/month)	Poor Households	60	(46)	32	(44)	27	(52)	70	(53)	34	(47)	35	(67)
		Non-Poor Households	71	(54)	41	(56)	25	(48)	61	(47)	39	(53)	17	(33)
		Total	*131*	*(100)*	*73*	*(100)*	*52*	*(100)*	*131*	*(100)*	*73*	*(100)*	*52*	*(100)*
	International poverty line (US$ 1/AE/day)[b]	Poor Households	65	(50)	34	(47)	30	(58)	86	(66)	46	(63)	39	(75)
		Non-Poor Households	66	(50)	39	(53)	22	(42)	45	(34)	27	(37)	13	(25)
		Total	*131*	*(100)*	*73*	*(100)*	*52*	*(100)*	*131*	*(100)*	*73*	*(100)*	*52*	*(100)*
Livestock Poverty	Livestock poverty threshold is 4.5 TLU[c]	Poor Households	46	(35)	24	(33)	19	(37)	52	(40)	27	(37)	22	(42)
		Non-Poor Households	85	(65)	49	(67)	33	(63)	79	(60)	46	(63)	30	(58)
		Total	*131*	*(100)*	*73*	*(100)*	*52*	*(100)*	*131*	*(100)*	*73*	*(100)*	*52*	*(100)*

[a]The concept of adult equivalent (AE) is based on differences in human nutrition requirements according to age, where; <4, 5-14 and > 15 years of age are equivalent to 0.24, 0.65 and 1 AE respectively
[b]The conversion rate used are US$ 1=KES 63.20 (June 30, 2008) and US$ 1=KES 73.98 (June 30, 2009). Source: www.oanda.com
[c]Tropical Livestock Unit (TLU)
[d]Excludes six households that were previously enrolled but reported to have dropped out of the OOC

Figure 6.8a and b shows the distribution of all the surveyed households based on the combination of cash income earned and livestock holdings in 2008 and 2009 respectively.

Figure 6.8 The distribution of OOC (blue colour) and non-OOC (red colour) households based on a combination of cash income (US$/AE/day) and livestock holdings (TLU/AE) taken as a proxy for wealth/poverty status for **A**. (2008) and **B** (2009).
Data source; Lead authors' survey
The four livelihoods groups are represented by the letters in capital: **P** (in red) represents the poor group; W represents the wealthy group; and **M1** and **M2** represent the two middle groups respectively.



The analysis of the data underlying Figure 6.8 shows that in 2008, 35% and 34% of all the households in the wealthy (W) and poor (P) groups respectively. About a third of the household (31% in 2008) were split between the two middle wealth groups, with M1 accounting for 15% and M2 for 16% respectively.

The changes in cash income and livestock holdings among all households in the MME are indicated in Figure 6.8b by the shifts in the median values compared to 2008. The median income dropped from US$ 1/AE/day in 2008 to 0.65 1/AE/day in 2009 while median livestock holdings dropped from 7TLU/AE in 2008 to about 5TLU/AE in 2009.

Household wealth inequality

Among all surveyed households, inequality is highest in cattle ownership (gini-index, GI=0.543), followed by shoat ownership (GI=0.500), cash income in 2009 (GI=0.424), and land ownership (GI=0.261) in that order (Table 6.5). Inequality in income (2009) and land ownership is less among OOC households than among non-OOC households, with significant differences in GI between the two groups for both variables.

In general the inequality in livestock ownership (both cattle and shoats) across all the surveyed households in the MME was high (GI for cattle = 0.543 and for shoats = 0.5). Although inequalities in livestock ownership (cattle and shoats) were higher among OOC households compared to non-OOC households, the difference was not statistically significant (Table 6.5).

Table 6.5 The Gini-index for gross cash income in 2009, land (among surveyed households and all households in OOC in 2010), cattle and shoats (sheep and goats) ownership in MME.
Data source; Lead authors' survey and the *Ol Purkel Ltd* for the data for 146 households enrolled in the OOC PES scheme

	Group	Estimate	SE[a]	95% LCL[b]	95% UCL[c]	DIG[d]	
Gross Income in 2009	Non-OOC Households	0.485	0.058	0.371	0.599		**P>|t|**
	OOC Households	0.352	0.043	0.266	0.438	(1.84)	0.071**
	All Households	0.424	0.035	0.354	0.494		
Land ownership (Surveyed Households)	Non-OOC Households	0.387	0.058	0.272	0.502		
	OOC Households	0.175	0.027	0.121	0.228	(6046.36)	0.000***
	All Households	0.261	0.030	0.202	0.319		
Land ownership (OOC)		0.153	0.045	0.065	0.241		
Cattle ownership (TLU)	Non-OOC Households	0.477	0.051	0.376	0.578		
	OOC Households	0.558	0.035	0.490	0.627	(-1.31)	0.196
	All Households	0.543	0.032	0.481	0.606		
Shoats ownership (TLU)	Non-OOC Households	0.475	0.055	0.365	0.585		
	OOC Households	0.504	0.040	0.424	0.583	(-0.41)	0.682
	All Households	0.500	0.033	0.435	0.564		

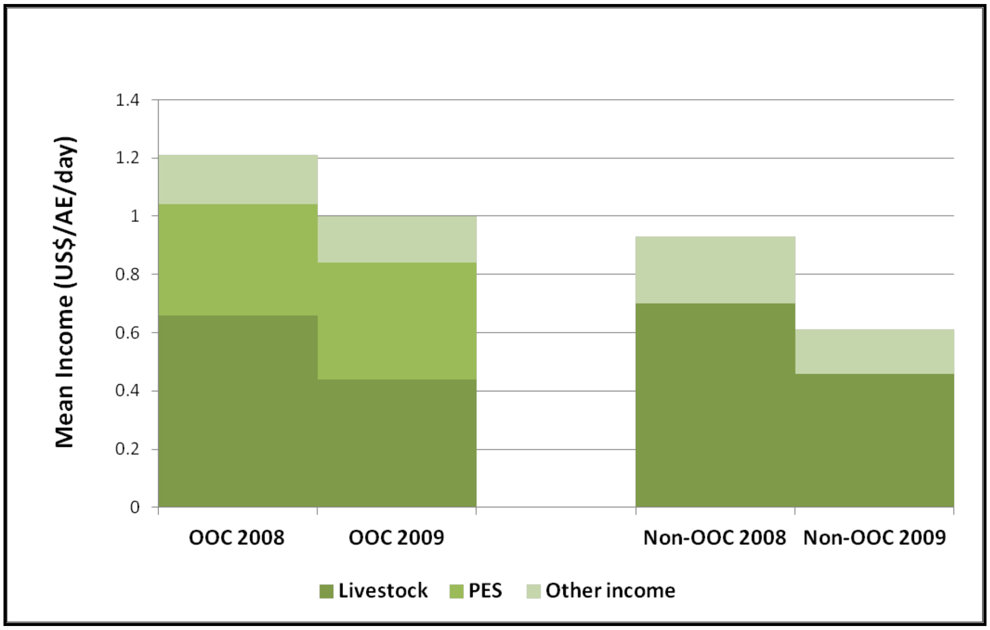
**P* < 0.1; ***P*<0.05; ****P*<0.01
[a]SE: Standard Error; [b]LCL: Lower Confidence Limit; [c]UCL: Upper Confidence Limit: [d]Differences in Gini-coefficient (degrees of freedom [DF] =57)

The effect of PES on cash income

The mean cash income for both OOC and non-OOC households decreased between 2008 and 2009 but the former's incomes were higher than the latter (Figure 6.9 and Table 6.6). In 2008, the household income for non-OOC households was 21% lower than OOC household and in 2009 this figure dropped to 13%.

Similar, per capita income for non-OOC households was 33% and 24% lower than for OOC household in 2008 and 2009 respectively, largely due to PES income. The lower percentage per capita values can be attributed to the high variation in household size among the surveyed households.

Figure 6.9 Mean per capita income (US$/AE/day) for OOC and non-OOC households in the MME in 2008 and 2009 disaggregated by livestock, PES and other income sources combined.
Data source; Lead author's survey.



As shown in Figure 6.9 and Table 6.6, livestock accounted for the highest share of household income, accounting for 55% in 2008 but declining to 46% in 2009 for OOC households. Among non-OOC, the share of livestock in the household income remained stable, accounting for 74% and 75% in 2008 and 2009 respectively (Table 6.6). PES income ranked second to livestock in both years. The share of PES in total household income increased from 30% in 2008 to 37% in 2009 (Table 6.6).

PES income had the lowest co-efficient of variation (CV) at the household level (27.39% in 2008 and 28.09% in 2009) and per capita (87.64 and 77.52% in 2008 and 2009 respectively) among the three main income sources considered although the CV of

PES per capita was much higher than that per household, as household size varied markedly (Table 6.6). In terms of gross annual income, OOC households had a lower CV at both the household level (63.25% in 2008 and 79.13% in 2009) and per capita (77.96% in 2008 and 74.86% in 2009) than non-OOC households (Table 6.6).

Table 6.6 Mean revenue (US$/HH/year and US$/AE/day), percentage of income and Coefficient of Variation (CV) from three sources of income for a sub-sample of households participating (enrolled in OOC: N=73), and not participating (N=45) in a PES scheme.
Data source: Lead author's survey.

Income Source		Year	Income		CV	
			Mean (US$) HH/yr	Percentage (%)	HH ($/HH/yr)	Per capita ($/AE/day)
Livestock	OOC	2008	3182.33	(54.62)	94.20	109.19
		2009	2282.64	(45.56)	152.06	124.95
	Non-OOC	2008	2843.72	(74.09)	83.86	102.96
		2009	1833.11	(73.09)	129.01	131.94
PES Income	OOC	2008	1,725.42	(29.61)	27.39	87.64
		2009	1857.74	(37.08)	28.09	77.52
Other Income Combined	OOC	2008	918.82	(15.77)	253.73	276.88
		2009	869.68	(17.36)	253.18	268.33
	Non-OOC	2008	994.30	(25.91)	183.09	180.49
		2009	675.08	(26.91)	132.21	156.75
Total	OOC	2008	5,826.58	100	63.25	77.96
		2009	5010.06	100	79.13	74.86
	Non-OOC	2008	3,838.03	100	78.09	93.04
		2009	2508.19	100	113.10	113.81

The effects of PES on expenditure choices

Our results show that basic needs accounted for the highest per capita PES expenses, averaging US$75 in 2009 (Table 6.7). Although expenses on education ranked second with a mean of US$40, it is still less than the combined expenses on livestock purchase, and veterinary services, which together amount to US$65 per capita (Table 6.7). In effect, the second highest per capita PES income is allocated to a combination of livestock related expenses.

Table 6.7 Per capita expenditure on PES income by OOC households on seven bundles of goods and services in 2009 (listed in decreasing order of mean values).
Data source: Lead author's survey.

	Household PES Expenditure in 2009	
Bundle of goods and services	**Mean (US$/AE/yr)**	**StdDev**
Basic needs expenses (food, cloths, etc)	74.74	85.01
Educational expenses (books, fees, etc)	40.22	60.42
Livestock veterinary expenses	34.81	57.23
Purchase of livestock (cattle, sheep, goats)	29.60	42.51
Human health expenses (drugs, hospital fee)	24.34	36.44
Purchase of hay/lease of land for grazing rights	1.84	9.97
Purchase of water for domestic consumption	0.79	3.62

Discussion

The outcome of land privatization and sub-division

Around the year 2000, as land sub-division was ongoing in the MME, uncertainty prevailed regarding which of the following five possible livelihood options would be adopted by Maasai landowners: "intensification" of livestock production, tourism, small-scale cultivation, large-scale cultivation, and land sales (Lamprey and Reid, 2004). There were concerns that land sub-division would curtail wildlife tourism if the process resulted in increase in land area under crop cultivation (Seno and Shaw, 2002, Kepher-Gona, 2006). In a survey carried out before the sub-division process was initiated in the MME, 82% and 52% of landowners speculated that they would use their land for livestock and for crop cultivation respectively. Only 27% mentioned tourism as a potential post sub-division land use option (Seno and Shaw, 2002). Contrary to these expectations, an overwhelming majority of pastoral landowners in the MME are opting to allocate their land for wildlife tourism under arrangements that to an extent, puts some constraints on pastoralism.

We argue that the establishment of clearly defined property rights to land (Norton-Griffiths, 1996, Swallow and Bromley, 1995) was one critical factor underlying the change of attitude among landowners to adopt a wildlife tourism based PES scheme in the Olare Orok Conservancy. First, land-subdivision and titling increased household land tenure security for those allocated land by transferring the property rights to land from collective to individual (or corporate) ownership (Mwangi, 2007a, Kabubo-Mariara, 2005). This in effect shifts the locus at which land rents (from livestock, wildlife, or agriculture, etc) are captured from community institutions where they are prone to inequitable distribution (Thompson and Homewood, 2002) to the household level, thus reducing distributional inequities in wildlife tourism revenues (Norton-Griffiths, 2007a).

Second, a section of landowners in the OOC were empowered by their ownership to land and could pull out of the proposed Conservancy if they were not satisfied with the prevailing conditions. As documented by Sorlie (2008), the local elites and the tourism partners depended on the cooperation of landowners and were in many circumstances forced to take landowners interests into account when designing the Conservancy model. It is apparent that once they had land tenure secured, landowners could then choose to invest different alternative land production uses. The challenge for landowners that enrolled in the OOC is that while in theory, they can withdraw from the conservancy; in practice this is complicated because of the challenge of access to the land and the cost of avoiding livestock infractions on conservancy land. Consequently, the only choice for a landowner that is dissatisfied with the conservancy is to sell their land. The risk is that such a situation may pave way for land dispossession through coercion and pressure.

Pastoralism and wildlife tourism in the MME both require open rangelands. Maintaining pastoral practices or adopting wildlife tourism both demand that landowner's co-ordinate and plan their land use collectively. This process may entail reverting to joint and collective management of land through re-aggregation of individual parcels to ensure the mobility of livestock and wildlife in a fluid ASAL ecology characterized by high spatial and temporal variability of key resources such as pasture and water (Niamir-Fuller, 1999, Hobbs et al., 2008a). The need to collectively provide large areas for tourism game viewing made the formation of conservancies and the adoption of PES through collective action desirable (Swallow and Meinzen-Dick, 2009).

Implications for livelihoods, poverty and inequality

In terms of livelihoods, our analysis shows that livestock remains the mainstay of household cash income in MME, supporting similar conclusions from other studies in the area (Thompson et al., 2009, Seno and Shaw, 2002). However, our results also show that wildlife tourism through PES, which ranks second to livestock in terms of the proportional contribution to household gross cash income, is an invaluable source of income diversification (Homewood et al., 2009b, Kristjanson et al., 2002), particularly during periods of severe droughts such as 2008-2009 (Osano, 2011).

Although we did not estimate the opportunity cost of landowners' involvement in the OOC PES scheme, the low dropout rate observed, coupled with the high acceptance rate of the 15-year contracts, suggests that the majority of landowners view the benefits as outweighing the opportunity costs. At least in theory, since participation is voluntary, landowners would presumably remain in a PES scheme if it does not make them worse off (Bulte et al., 2008b, Pagiola et al., 2005). Also, although small-scale and commercial agriculture might provide other alternative land use for the landowners, the Olare Orok area is not as agriculturally productive as are other parts of the MME (Lamprey & Reid 2004), and would also attract high costs in terms of wildlife damages to crops.

A desirable feature of the OOC PES model, which is based on fixed leases, is that cash income is regular and predictable, unlike other sources of household income such as livestock and cropping, which are highly seasonally variable (Bosire et al., 2012). However, households in the OOC are faced with inevitable trade-offs because as a "land-diversion" PES scheme (Zilberman et al., 2008), which excludes settlements and limits pastoral livestock grazing, the scheme can negatively affect livestock production.

Although the resettlement process was voluntary, and the costs were largely borne by the tourism partners and local elites (Sorlie, 2008), it is apparent that the creation of the OOC also led to involuntary displacement of migrant households that had settled in the Olare Orok area, but were not allocated land elsewhere. Some of these families that were displaced involuntarily have resettled on land inside the Group Ranches that are yet to be sub-divided and still retained under communal management (Courtney, 2009).

Moreover, since the creation of Conservancies could have led to the displacement of both people and livestock, with different levels of restrictions to livestock grazing among the eight current Conservancies, it is clear that there is increased pressure exerted on grazing lands, if the livestock populations remain stable or increase. This is because it is unlikely that all the landowners who relocated their settlements from Olare Orok had adequate alternative land elsewhere to settle and graze their livestock. There is a need for studies to look into what impact the relocation of settlements and displacement of livestock herds has had on the intensification of land use in the re-settlement sites.

Clearly, the livestock grazing restrictions within the OOC (and other Conservancies), by limiting pastoralism and pasture available for landowners, transfer pressure to communal pastoral areas (Bosire et al., 2012) and to the Maasai Mara National Reserve where illegal livestock grazing has increased (Ogutu et al., 2011). The constriction of livestock grazing areas can potentially to amplify the vulnerability of the pastoralists to the recurrent climatic extremes such as severe droughts (Western and Manzolillo Nightingale, 2003, Galvin, 2009).

Our analysis reveals three main implications in terms of income poverty. First, the high income poverty rate of 63% in the MME suggests that there is potential for poor household to fall within zones targeted for PES intervention. However, spatial targeting that limits the scheme to people with land within the Olare Orok area only excludes the poor outside the target zone. Second, the PES rates of US$ 40/ha/year in the OOC are competitive against expected land rents from agriculture and livestock within MME in areas receiving a mean annual rainfall of 600 mm and 900 mm, respectively (Norton-Griffiths and Said, 2010, Norton-Griffiths et al., 2008), and are much higher than the estimated returns of US$ 2/ha/year from pastoralism alone or pastoralism and ecotourism combined (Thompson, 2005). There is however considerable scope to increase the land holders' wildlife tourism revenue, up to US$ 60-65/ha or higher, if the PES scheme is supplemented by an equitable system of park revenue sharing. For instance, the MMNR visitor entrance fees alone is estimated to be capable of generating US$ 5.5 million annually (equivalent to US$ 100/ha) (Walpole and Leader-Williams, 2001) which can go a long way in the provision of additional wildlife income to complement the existing PES schemes in the MME Conservancies.

Third, the magnitude of the OOC PES cash transfer is, on average, sufficient to close the estimated maximum poverty gap of 20% in MME; on average, a poor person in a locality with a 20% poverty gap will require an additional monthly income of KES 248 to move above the rural poverty line of KES 1,239. The OOC PES payments provide an annual rent of KES 3,750 per hectare (in 2011), which translates to a monthly rent of KES 313 per hectare. The mean land enrolment among some 146 households in the OOC is 56.75 ha, so the OOC PES transfers, on average, a monthly per capita cash income of KES 17,734, which is of a magnitude far above what is required to lift all households found in locations with a poverty gap of 20% above the rural poverty line.

This conclusion should be taken cautiously though as the poverty gap is an average which conceals the variation inherent in the depth of poverty, and thus only provides a crude indication of PES impact on poverty based on the magnitude of household cash transfers. It can however be a useful starting in the design and development of "Payments for Ecosystem Services and Poverty Alleviation (PESPA)" programs (Rodriguez et al., 2011b).

The implications of PES in terms of wealth inequalities are illuminating. As expected, there is a very high disparity in livestock (cattle, sheep and goat) ownership among the surveyed households. This is unsurprising as inequality in household livestock ownership is a well-known phenomenon among Maasai pastoralists (Radeny et al., 2007, Kabubo-Mariara, 2005, Homewood et al., 2009c). Surprisingly, our results show that the level of inequality in terms of land ownership is, in general, low among all surveyed households, which may be attributed to a relatively equitable process of land sub-division. Although, the inequality in landownership in OOC is low, there are a few cases of households that were allocated parcels considerably larger than the average. An example is the local Chief Kipeen ole Saiyalel, who played a leading and critically instrumental role in the establishment of the OOC (Sorlie, 2008). He is said to have been allocated 1000 ha compared to the average of 60 ha for the majority of landowners (Courtney 2009).

In general, the level of land inequality in land ownership in the MME appears to be low in contrast to other pastoral rangelands in southern Kenya where land privatisation and sub-division were highly inequitable (Mwangi, 2007b, Rutten, 1992, Galaty, 1999). Since PES payments are based on land holdings and the rates ($/ha/year) are uniform across all OOC household, it is not expected that the OOC PES scheme may exacerbate existing income disparities among the participants. The only exception relate to the households on whose land the campsites are located because they are paid an additional bed-night fee of US$8 per visitor per night, on top of their annual PES income.

Our findings show that of the three income sources considered in our study (livestock, wildlife tourism PES and other income sources combined), PES payments recorded the lowest co-efficient of variation (CV), suggesting that it was the most equitable, suggesting that the OOC PES scheme has an income inequality reduction effect among the participants. On the other hand, it also serves to widen the income disparity between participants and non-participating households because the difference in income between these two groups was found to be statistically significant

A point of interest in PES schemes is how the recipients spend the PES income, and the implication of their expenditures on household welfare and poverty (Grieg-Gran et al., 2005, Miranda et al., 2003). Bulte et al. (2008), for example, suggest that the Maasai will invest PES income in expanding livestock herds. Others suggests that livestock herders participating in PES schemes that enhance interaction between wildlife and livestock would use the PES income to pay for veterinary services (Horan et al., 2008). Our findings show that the four leading bundles of goods and services to which households allocated their PES income were basic needs, livestock (combining livestock and hay purchases together with veterinary services) and education in that order.

The implication for biodiversity and wildlife conservation

A critical concern in the PES schemes is their impact on biodiversity conservation (Ferraro and Pattanayak, 2006) and "*additionality*"; that is, do PES programs supply environmental services that would not have occurred in their absence (Pattanayak et al., 2010). Although assessment of the biodiversity impact of the OOC PES scheme is not part of this study, there is a great expectation that the OOC and other conservancies will deliver on biodiversity conservation, and stall the current rapid decline in wildlife within the MME (Ogutu et al., 2011, Ottichilo et al., 2000) and in Kenya (Western et al., 2009). This expectation stems from the observation that wildlife losses in ASALs have been less where tourists visited than where they did not, and in cases where wildlife revenues went more clearly and transparently to landholders (group, communal or private) rather than to the central government, wildlife was either holding its own or perhaps even increasing (Norton-Griffiths and Said, 2010).

Clearly, the PES schemes have significant implications for biodiversity policy and practice but we highlight only two here. First, voluntary resettlement can lead to biodiversity conservation in abandoned land (Young, 2006) and the OOC and other Conservancies are already opening up wildlife corridors and dispersal areas around the MMNR, allowing for species dispersal movements between their wet and dry season concentration areas (Bhola et al., 2012). However, the displacement of settlements and livestock can also generate knock-on effects through leakages as household's crowd on communal lands (Bosire et al., 2012) and livestock encroach on the protected areas (Ogutu et al., 2011).

As Conservancies expand in the MME, each being operated independently, and having different levels of livestock grazing restrictions, the resultant leakages in both the communal lands and the Reserve may impact negatively on wildlife biodiversity. This is may be worsened by the fact that Conservancies are currently established opportunistically, in the absence of an ecosystem-wide management plan to guide land use planning and zonation (Bosire et al., 2012). It is therefore incumbent on the MME stakeholders (the MMNR and Conservancy management, tourism operators, conservation agencies and local communities.) to develop a holistic ecosystem management plan to integrate both the MMNR and the surrounding lands (Ervin et al., 2010). A process is currently underway to explore the formation of an umbrella institution that brings together all the Conservancies in the MME. A management plan developed jointly by the Conservancies can be integrated with the management plan for the MMNR (NCC & TCC, 2009).

Second, although the conservancies in the MME have evolved as private sector-led initiatives largely through partnerships between commercial tourist enterprises and pastoral landowners with minimal involvement of the government (Homewood, 2009), the emerging challenge, however, is the lack of a policy framework and/or statute that governs Conservancy establishment and operations (Bosire et al., 2012). In an effort to fill this policy gap, the Kenya Wildlife Service (KWS) has recently built upon the Draft Wildlife Bill (Republic of Kenya, 2011b) and developed a policy document, which, among other things, seeks to provide policy guidance on the establishment of Wildlife Conservancies (KWS, 2012). While this is a laudable effort, a careful scrutiny of the policy document shows that it places too much emphasis on regulation and enforcement with little appreciation of the role played by tourism in the establishment of Conservancies in pastoral lands. As an example, the policy document does not make any reference to the *Tourism Policy* or the *Tourism Act* 2011 yet wildlife tourism is the principal means of revenue generation in Conservancies (Bosire et al., 2012). The National Tourism Policy is explicit that a major constraint faced by the industry is the lack of harmonization between national policies on land-use, wildlife and tourism (RoK

2011: 10). It also appears the drafting of the document was not participatory and largely excluded pastoral landowners in whose land Conservancies are expected to be established.

Also, both the *Draft Wildlife Bill* and the policy propose stringent requirements for registration of Conservancies, which include the proposal to gazette the area set aside as a Conservancy. In practice, this requirement could discourage pastoral landowners from participation in Conservancies out of fear for land annexation by the Conservation NGOs and the state to expand protected areas. The PES mechanism is a voluntary mechanism which allows potential participants to join and leave at will (Wunder, 2005, Engel et al., 2008). Conservancy regulations that restrict landowner's voluntary exit from PES schemes will be a disincentive for their participation in conservation.

The implications for tourism

A key principle of sustainable tourism is that it should generate minimum negative livelihood disruptions especially for the poor (DFID, 1999, Ashley, 1995). As the PES scheme in OOC provides tour operators with exclusive control over the leased land (Carter et al., 2008), the restrictions on livestock grazing, which is the mainstay of the Maasai economy, is a key challenge for the sustainability of this scheme in the long run. While the conservancy management has responded to this challenge by allowing controlled rotational grazing, and opening up particular areas during drought periods (Osano, 2011), some of the tourists still do not want to see livestock within conservancies, as aptly captured in the following remark: "*Tourists don't come to see cattle, they want 'big cats', and big cats don't mix with cattle, even if the other plains game do*."[6] Educating tourists on the importance of livestock grazing to wildlife, and complementing wildlife tourism with cultural tourism are two options worth pursuing in this regard (Bosire et al., 2012).

[6] Cited from a presentation "The Olare Orok Conservancy and the structures which help it work" (n.d. on file with the author)

Separately, concerns have been raised regarding the sustainability of the tourism PES schemes due to the vulnerability of the Kenyan tourism industry to political, financial and economic shocks (Ikiara, 2001) plus the current threat posed by international terrorism, which may negatively affect households dependent on tourism based income (Kareithi, 2003). So far, the OOC PES scheme has addressed this concern in two ways. First, the commercial tourist partners have put in place a Contingency Fund to buffer the landowners from such stochastic events. To their credit, the tourism partners in OOC continued providing payments to landholders, even when tourism was on the verge of collapse and many camps had closed during the 2007-08 post-election violence in Kenya (Sorlie, 2008). In the long term, this can be complemented by, for example, insurance underwriting to cover risks resulting from tourism downturns (Bosire et al., 2012) or establishment of an endowment kitty.

Second, the individual tourism enterprises also share the risks among themselves such that should one or more partners withdraw from the OOC before expiry of the contracts, the remaining partners will still provide the payments to land owners until another partner is brought on board to fill in the gap (O'Meara, 2011).

Lastly, the OOC is able to lease land at the rate of US$ 40 per hectare per year which is higher relative to rates prevailing in other parts of Kenya of around US$ 10/ha/ year (Gichohi, 2003). This is because the OOC targets the high end tourism market and the charges range from US$ 600-1000 per tourist per night enabling the tourist partners in OOC to meet their financial obligations to landowners at about 60% occupancy rate. The high end tourism market segment is however small and already saturated (with about 40-50 high-end lodges present in Kenya) hence has limited expansion potential in the country (West Gate Conservancy, 2012). This poses a challenge to other emerging conservancies in the MME and elsewhere that may have to target the low end or middle level tourism market segments, and thus may not be able to match the OOC rates. This calls for the diversification of conservancy revenue generation from tourism to consider many possibilities, including tapping into PES for other ecosystem services other than wildlife tourism and recreation.

Complementary funding from government and public sources could augment tourism payments to landholders, especially if wildlife tourism PES is explicitly incorporated in national policies and statute. This will enable conservancies to tap into funds from both the wildlife and tourism sectors as is happening in Costa Rica where the national PSA PES program explicitly recognizes the provision of scenic beauty for recreation and ecotourism as one of the four environmental services provided by forest ecosystems, and benefits from government payments (Pagiola, 2008, Huberman, 2009). In Kenya, the current draft National Wildlife Bill (2011) lists payments for environmental services (PES) as one of the fiscal instruments for wildlife conservation and stipulates that communities neighbouring protected areas and contributing to the provision of environmental services will co-share the benefits (Republic of Kenya, 2011b).

Conclusion

Global policies seek to promote the linkages between poverty eradication and biodiversity conservation (Sachs et al., 2009) to ensure win-win outcomes (Timmer and Juma, 2005, Sanderson, 2005). Wildlife tourism is presented as having the potential for simultaneously promoting biodiversity conservation while also improving rural livelihoods and poverty reduction (DFID, 2002, DFID, 1999, Okello et al., 2009).

This paper contributes to the broad question regarding the role of wildlife tourism in poverty reduction among pastoral communities in Africa's pastoral drylands by focusing on the Maasai Mara Ecosystem (MME), a prime area for wildlife and a leading wildlife tourism destination in East Africa. More specifically, it examines the effects of wildlife-based tourism implemented through Payments for Ecosystem Services (PES) on household poverty, wealth inequality and livelihoods within the context of changing land tenure from collective to individual property rights to land. It uses the case of Olare Orok Conservancy (OOC), which has adopted a PES scheme in which Maasai landowners have agreed to voluntary resettlement and exclusion of livestock grazing by leasing their land for wildlife tourism in return for cash payments by a coalition of tourism operators. In doing so, the paper seeks to unravel the puzzle why the Maasai landowners in OOC chose to keep lions on their sub-divided lands instead of livestock (which is still retained on land elsewhere).

We draw five main conclusions from our analysis. First, that land privatization and sub-division in the MME is a double-edged sword. On the one hand, there are legitimate fears, based on experiences in other parts of Maasailand, that by promoting fragmentation, land-subdivision could lead to land uses that are inimical to pastoralism and wildlife conservation. On the other hand, our analysis shows that sub-division and titling of land appears to have provided individual households that benefitted from the land allocation with the exclusive rights to land, enhancing their security of tenure while increasing benefit sharing from the wildlife based tourism among the OOC landowners.

Second, the OOC PES scheme impacts on poverty in two ways. One, the high income poverty rate in the MME means that the majority of the potential participants are likely to be poor hence may therefore benefit from the scheme. However, because of spatial targeting, participation is restricted to households with land in the Olare Orok area. Two, the magnitude of the PES cash transfer is, on average, sufficient to lift all the poor households in the study area above the Kenyan rural poverty line. This conclusion will have to be assessed on account of the opportunity costs to landowners of participation, but which was not within the scope of this study.

Third, PES is an invaluable source of income diversification providing the second highest contribution to household cash income during periods of severe droughts when livestock income may decline. Furthermore, the PES income is even more valuable because it is regular and predictable, unlike other income sources which are highly variable and less predictable, and it goes towards paying for basic needs including purchases of food and clothes, and towards maintaining and expanding livestock herds.

Finally, although PES income is the most equitable of all the income sources among the OOC households, and thus lessens the overall household cash income inequality among the PES participants. It also generates a negative derived effect by exacerbating income inequality between participants and non-participants. The exact extent of this effect is still unclear, and requires further studies, to also move beyond OOC alone and consider all the conservancies operational in the MME.

Although our analysis suggests that PES contributes towards the improvement of pastoral livelihoods, and has a high potential for wildlife conservation, several issues concerning their implementation require policy and programmatic attention. First, from a poverty reduction and wealth inequality perspective, an ideal situation would be where the landholders are capacitated to run the tourism enterprises themselves but in the current arrangement, the tourism revenues are still disproportionately skewed towards the tourist enterprises. In addition, the spatial targeting of the PES program means that it excludes both landowners with land outside the Olare Orok area, and also the landless that may be made worse–off by restricting their participation, while at the same time excluding them from the land and pasture. Households with land outside the Olare Orok area may still have an opportunity to join and benefit from other conservancies that have developed in the MME

Second, there is need to promote integrated tourism and wildlife policies that support rather than exclude pastoralism. Tour operators should be enlisted to show tourists that at low to intermediate stocking levels livestock can co-exist with wildlife and promote biodiversity conservation. In addition, flexibility in conservancy rules are required particularly during drought as the constriction of areas available for livestock grazing due to conservancy establishment may render pastoral landowners more vulnerable to climatic variability.

CHAPTER 7: CONCLUSIONS

Summary of the chapter findings

The summary findings from the substantive manuscript chapters are as follows;

Chapter 3 explored the potential for PES to alleviate income poverty in the pastoral ASALs in Kenya in relation to changes in biodiversity (using wildlife biomass as a proxy), selected ecosystem services, land use, demography and climate variability, and came up with the following key findings.

- A large expansion of cropland occurred in Kenyan ASALs between the 1970s and mid-2000s. This expansion was at the expense of both wildlife and livestock production suggesting a mixed outcome for provisioning services and indicating loss of biodiversity based on declining wildlife populations.
- The supply of habitat services on private and communal land increased over the whole of Kenyan ASAL areas and within the Maasailand region where the supply of tourism cultural services also increased between 1990 and 2010.
- There is a potential for PES to benefit the poor semi-nomadic populations because relatively low amounts of cash transfers per hectare are required to close the poverty gap in ASAL area, and based on the existing correlation between areas rich in wildlife and areas recording high rates of poverty in ASAL. The investments in PES would have to be generated from the government and public sector given the limited operations of the private sector in the tourism industry.
- There is a recorded decline in precipitation and an increase in temperature over the last 40 years. It is projected that temperature trends will continue into the future, but there remains uncertainty regarding the future rainfall trend. Continued changes in these two climatic variables are expected to in the short-term and will alter weather patterns thereby affecting pastoral livelihoods.

Chapter 4 assessed the role of PES in adaptation to climate change and poverty among pastoral communities. It presented a conceptual framework linking PES to ecosystem based adaptation (EBA) to climate change. The key findings in the chapter include;

- Local droughts in the two study sites in Athi-Kaputie and Maasai Mara Ecosystem have increased in severity and frequency over the last several decades.
- PES serves as a “safety-net” that buffer participating pastoral households against the loss of livestock income during drought periods. PES can therefore cushion households against climatic shocks.
- PES serves multiple roles in Ecosystem based Adaptation (EBA). These include enhancing access to capital and technology, modifying local level institutions relevant to adaptation, and creating vertical and horizontal linkages for pastoral communities, all of which contribute to towards shaping the adaptive capacity of local communities in both positive and negative ways.

Chapter 5 examined the effects of PES on household poverty, wealth inequality and livelihoods in the Wildlife Lease Program (WLP), a publicly funded PES scheme whereby pastoral land users living within the wildlife dispersal corridor adjacent to Nairobi National Park are paid to refrain from cultivation, land sales and sub-division, and to allow wildlife on their private land. The findings here show that;

- Although it has a bias towards for poor families occupying areas with low quality pasture, participation in the WLP is not strictly pro-poor and is skewed towards non-poor households; even among the households in the WLP, non-poor households have a higher intensity of participation than poor households.
- PES serves as an invaluable source of income diversification which contributes a substantial proportion of participating household’s cash income during drought period when the share of livestock income may decline.
- The PES effect on income inequality is two pronged. On the one hand, PES was found to have an income inequality reduction effect among participants. On the other hand, it also serves to widen the income inequality between the participants and non-participants.
- The magnitude of PES income, not accounting for the opportunity costs of participation, is on average sufficient to close the poverty gap in the Athi-Kaputie Plains.

Chapter 6 examined the effects of PES on household poverty, wealth inequality and livelihoods in Olare Orok Conservancy which is a private sector led PES scheme in the Maasai Mara Ecosystem. In the OOC, Maasai landowners have agreed to voluntary resettlement and exclusion of livestock grazing from their aggregated land parcels which are set aside for wildlife tourism, in return for cash payments by a coalition of tourism operators. The findings show that;

- Land privatization appears to have provided pastoral households allocated land in the OOC with exclusive rights to land, enhancing their security of tenure, and enabling their participation in the PES.
- PES serves as an invaluable source of income diversification accounting for a large share of household income during period of drought when the share of livestock income may decline.
- The PES effect on income inequality is two pronged. On the one hand, PES was found to have an income inequality reduction effect among participants. On the other hand, it also serves to widen the income inequality between the participants and non-participants.
- The magnitude of PES income, not accounting for the opportunity costs of participation, is on average sufficient to close the poverty gap in the Maasai Mara Ecosystem.
- PES may generate leakages, by displacing settlements to the pastoral commons, and livestock grazing to the Maasai Mara National Reserve.

Situating the study findings in light of the debate on the ethical issues concerning PES and the Neo-Classical Economic framework

Payments for ecosystem services (PES) are becoming a popular mechanism for the conservation and management of ecosystem services that is currently being promoted by global development and conservation institutions. These include the multilateral agencies such as the Organization for Economic Cooperation and Development (OECD), the World Bank and the United Nations, the Conservation organizations such as The Nature Conservancy (TNC), the World Resources Institute (WRI), the World

Conservation Union (IUCN), and the World Wildlife Fund (WWF), and private sector corporate businesses entities spearheaded by the World Business Council for Sustainable Development (WBCSD). The World Bank acting as an agency of the Global Environment Facility (GEF) and the TNC have both provided funding to the Wildlife Lease Program PES scheme (Chapter 5) while the OOC PES scheme is funded by the private sector corporate companies in the tourism industry.

Critics see PES as part of the process of expansion of the neo-liberal project, which includes within it, the corporatization of environmental policy (McAfee, 2012, Arsel and Büscher, 2012). This perspective is informed by the perception that the PES approach is based on the notion that nature can be commoditized, and that the use of markets can be relied upon to provide a solution to the current environmental crisis facing the anthropocene (McAfee and Shapiro, 2010). In this context, the institutions leading and promoting ecosystem service projects around the world, are responding to an increasing trend of supporting the use of market-based instruments to protect ecosystem services and biodiversity.

A key feature of this neo-liberalization trend is the proliferation of global studies and policy processes seeking to entrench the concept of ecosystem services (ES) and PES mechanism in environmental policy (Gómez-Baggethun et al., 2010), that are especially prominent in the emerging science-policy interface for biodiversity (Perrings et al., 2011). These include the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2005b), The Economics of Ecosystem and Biodiversity (TEEB, 2009), and the Intergovernmental Platform on Biodiversity and Ecosystem Services (Larigauderie and Mooney, 2010).

Inherent in the ongoing debate about the *pros* and *cons* of PES application in different socio-ecological contexts is the moral, ethical and philosophical issues that emerges from the PES approach (Luck et al., 2012), and the concerns regarding whether PES interventions, in their different variants, can be sustainable financially, socially and culturally in the long term (van Noordwijk et al., 2012, Kinzig et al., 2011).

The effectiveness and practical application of PES has been questioned on the basis of its ethical and moral philosophy that stress a vision of nature "*where countable, measurable*, *and monetary aspects dominate*" (Martin et al., 2008, Swart et al., 2003) and which presupposes that nature's role is limited to promoting human welfare. Indeed, the very notion of ecosystem services connotes an anthropocentric view of nature that does not take into account the well-being of other non-human species. As argued by Brown (2008), this is problematic because the ecosystem services concept ignores the well-being of non-human species in environmental policy, which should be conceptualized around the notion of the commonwealth of life (Brown, 2008). Serious concerns have been raised regarding the ethical issues in PES. These include the following;

Economic metaphor

An economic framing underlies the ecosystem services concept, which is based on a metaphor that describes ecosystems as (natural) capital and ecosystem functions as (ecosystem) services, potentially favoring the expansion of the rationality of profit calculus to the environmental domain (Gómez-Baggethun and Ruiz-Perez, 2011). The dominance of this metaphor in policy fails to adequately consider the fact that nature should also be recognized because of its intrinsic moral value, which is inappropriately reduced when nature is subjected to simple economic measures (Sagoff, 2011).

In the OOC for example (Chapter 6), the implementation of PES led to the imposition of livestock grazing limitations inside the Conservancy. The restriction to livestock grazing undermines the traditional semi-nomadic pastoral livelihood practices of the Maasai community. Although the Maasai families enrolled in the conservancy do gain financially from the PES program, this gain is at the expense of having to give up large areas of critical dry-season pastures which are now set aside exclusively during certain seasons for wildlife tourism and for the enjoyment of wealthy mostly international tourists who are able and willing to pay for the access to the Conservancy. Nature is thus being set aside for people who have the money to pay for it for recreational purposes but exclude the locals who need the space to meet their basic livelihood needs.

Worldwide, many traditional rural communities such as the Maasai have had to grapple with the logic presented by this economic metaphor of PES. In rejecting PES for example, a representative of an indigenous Mexican community noted;

> "Our in-depth analysis of the appropriateness of PES for the NTNP (*Nevado de Toluca National Park*) revealed that the concept of PES has some inherent contradictions. The PES approach conceives of the environment as a discrete group of services to which an economic value can be allocated; in an ambiguous situation in which supply and demand rarely share the same geographical space. Environmental service do not have a market that allows free competition and price fixing; buyers do not have options and the seller is constrained to a limited group of institutions; moreover, information between buyers and sellers is imperfect and it is not always clear what is being bought or sold. In this context, the value that is often allocated does not necessarily reflect the real value of benefits received" (Franco-Maass et al., 2008).

Indeed, if one considers the full spectrum of the social, cultural and economic costs to the local pastoral landholders, then the payments provided to the pastoral landholders may be insufficient to compensate for this full spectrum of costs.

Commodification and monetary valuation

PES is viewed as constituting an emerging institutions which attempt to reconfigure human–environment interactions by promoting the conservation of ecosystem services through their commodification, where traditionally, ecosystem services, were issued by institutions (protected areas and conservation policies), which have set aside some parts of nature from markets (Corbera et al., 2007b). PES is thus increasingly being seen as part of a neo-liberal project that promotes commodification of nature (Kosoy and Corbera, 2009).

One aspect of the commodification process is the valuation of nature and ecosystem services to determine how much they are worth in dollar terms (World Bank, 2005).
In most PES programs, payment rates are set on the basis of the valuation of ecosystem services, and several ethical issues arise from this. The monetary valuation of ecosystem

services is largely as anthropocentric because it views nature from a utilitarian perspective. Furthermore, it ignores the fact that not all components of nature can be valued because most of nature's components are intangible and cannot easily be equated to a dollar value. Consequently, most PES programs rely on observable proxies for ecosystem services, such as land use.

In a majority of PES programs, including both the WLP and the OOC PES schemes, payments are made for a specific land use, which is assumed to generate the desired ecosystem service. This is done without a clear evidence of the linkages between the land-use promoted and the ecosystem services desired and sometimes resulting in the promotion of one type of land use promoting a single ecosystem service while ignoring other land use types that equally generate ecosystem services that are valuable to the local community who are unable to pay for these ecosystem services. In this situation, PES can be dominated by commercial profit motives which may result in sub-optimal ecosystem health outcomes. This is the case for the OOC where initially, livestock grazing was totally excluded from the conservancy arguably creating an under grazing problem.

Furthermore, because of the lack of information that demonstrates the biophysical link between land uses and ecosystem service outcomes, as well as lack of suitable methods for measuring and monitoring outcomes, many current PES programs are based on beliefs that are not scientifically proven (Meijerink, 2008). It is therefore;

> "...quite likely that, at least in some areas, PES programs are promoting the wrong land uses for the ES they desire –for example, by increasing forest cover in areas with water deficits" (Wunder et al., 2008: 846).

There are also challenges with valuation methods which can be problematic especially in the context of information and power asymmetry between potential ES users and ES providers. In a contingent valuation (CV), which is a frequently used tool in the monetary valuation of ecosystem services, respondents can overstate their opportunity costs with the expectation of being paid more than they would need to conserve ecosystem services. For example, in a CV study that was conducted prior to the establishment of the WLP (see Chapter 5), to assess how much payment the pastoral landowners were willing to accept as compensation for losses incurred by the presence of wildlife on their land, the landowners reportedly asked for an annual compensation of US$ 1,260/ha/year (US$ 920/acre/year) (Mwangi and Warinda, 1999). It turned out that this was a grossly overstated figure as the majority of the same pastoral landowners are now enrolled in the WLP and are accepting a much lower payment of only US$ 10/ha/year (Chapter 5).

Another aspect of the Commodification is that it is strongly linked to the nature of property rights that exists for natural resources. The Neoliberal agenda emphasize the privatization of natural resources as a *sin-qua-non* for their effective management. Free market environmentalists for example argue that complete private property rights can help conserve natural resources such as wildlife. They advance the argument that private ownership promotes more efficient use where the owners of natural resources are free to use them in ways that grant them the highest income (Smith, 1981, Goodman and Stroup, 1991). It is evident in this study that the nature of property rights to land and wildlife resources are very critical to the implementation of the two PES schemes assessed.

In the Kenyan ASAL, PES has emerged in the context of land privatization and sub-division, which has been an ongoing process since the early 1970s (Galaty, 1994a). The process involved land titling and land-use planning with the hope that registered title will enable pastoral land owners to have sufficient level of security to invest in land for increased productivity, resource protection from over-exploitation, and encourage financial improvements through credit provision (Lane, 1998). Land titling can be advantageous because it may provide protection to pastoralists against land appropriation, but it could also be problematic because it may lead to landowners adopt land management practices that hinder pastoralists seasonal mobility across the landscape or lead to a situation where individual landowners may deny livestock herders access to their land (*ibid*.).

Despite the fact that in both the Athi-Kaputiei Plains and Maasai Mara Ecosystem, the privatization, sub-division and land titling has enabled pastoral landowners to participate in PES schemes (Chapter 5 and 6), many pastoralists view land subdivision as a harsh trade-off between the need for security of tenure on the one hand and the loss of social networks, livestock productivity, and drought resilience on the other. The existing government land policy in Kenya favors subdivision of rangelands in the interest of individual enterprises and the maximization/intensification of livestock production, rather than collaborative arrangements and joint wildlife–livestock enterprises for ecological viability (Curtin and Western, 2008).

Another property rights challenge to PES implementation in the ASALs, is the existing multiple land tenure systems. This can undermine the effective implementation of PES which can be weakened by among other factors, the pervasive disconnect between customary and statutory land rights. This lack of a well defined property rights and land tenure regimes has raised concerns that PES schemes in rangelands can open up opportunities for powerful forces, corporations and wealthy people to appropriate pastoral land under the guise of promoting conservation – the so called “green grabs’ (Fairhead et al., 2012).

The property rights to land (tenure, ownership and access) are particularly important in land-based PES programs because here payments are made to a landowner for a specified land use, and the landowner is expected to demonstrate legal ownership of the land. In both the WLP (Chapter 5) and the OOC (Chapter 6), the possession of a title deed is a requirement for enrolment to be able to demonstrate that one possesses the authority, ability, and willingness to restrict access and use of resources on contracted land.

The property rights to wildlife in Kenya are retained by the state, and not individuals and communities (Kameri-Mbote, 2002). In Kenya, large populations of wildlife are found in communal pastoral rangelands outside protected areas where their populations are declining, and arguments in favor of privatization as a way to secure the future of wildlife on private land have been put forth (Norton-Griffiths, 2007a, Norton-Griffiths, 2007b). Proponents of wildlife privatization point to South Africa and Namibia as two countries with successful models of conservation based on wildlife privatization (Muir-Leresche and Nelson, 2000, Muir-Leresche and Nelson, 2001). In Kenya, ethical concerns are one of the reasons advanced by the animal welfare groups such as the International Fund for Animal Welfare (IFAW) in their opposition to policies that allow wildlife privatization.

There are also several other reasons why the privatization of common pool resources (CPRs) including wildlife and pastoral lands are not favored by some. As a fugitive resource for example, it is challenging to privatize some wildlife resources such as the seasonally migratory species (Naughton-Treves and Sanderson, 1995, Tisdell, 2004). It is also argued that even in the cases where complete property rights to CPRs exists, and markets are efficient, private property rights do not always result in resource conservation; under certain circumstances, for example, uncertainty about resource availability due to unpredictable biological systems, economic pressures, and long term horizons, people can and will over-exploit resources they own privately (Acheson, 2006, Tisdell, 2004). Thus, privatization may not be the *sin-qua-non* that is presented. Indeed, as Brown (2008: 109) has pointed out;

> "Within the mainstream conception of property rights as advocated by the 'privatization' solution to environmental degradation there is no place for the duty, though no explicit prohibition either, to conserve natural communities" (Brown, 2008).

Socio-cultural impact and changes in motivation

The use of market instruments in the application of the ecosystem service concept raises ethical questions about their socio-cultural impacts. This is especially pertinent in the context of rural or indigenous communities. The socio-cultural impacts of PES which may be either positive or negative may include long-term changes in quality of life, independence, attitudes or belief systems, culture, security, the empowerment of women, community identity, or other changes in behavior and motivations for conserving nature (Luck et al., 2012). The fear that PES can lead to the loss of cultural motivations for conservation among the recipient communities has led to a cautionary note pointing out the need for a balance of cultural and incentive based approaches to conservation (Martin et al., 2008).

As an initiative to create incentives for pro-conservation behavior among participants, it is clear that PES may induce behavioral change but how this leads to changes in the motivations for nature conservation and ecosystem management remains unclear. As already pointed out in Chapter 2, there are many alternative perspectives and reasons why different groups of people engage in PES and these engagements are driven by different motivations, desires and expectations.

The utilitarian framing of PES in particular may crowd out moral and ethical motivations for nature conservation and stewardship of natural ecosystems (Vatn, 2010) in several ways. First, if the motivation for land owners to engage in conservation is promoted purely through monetary payments and incentives, it may become difficult for them to maintain this motivation in the absence of money, or to return to non-monetary motivation of cultural, aesthetic and spiritual motivations. This is of major concern because most of the existing PES schemes, including the two analyzed in this thesis lack long term funding mechanisms and financial sustainability is not guaranteed. Consequently, there exist uncertainties whether landowners will maintain the PES promoted practices in the absence of payments. This is a key research question for further exploration in relation to the WLP and the OOC PES schemes.

Equity implications

Poverty which is at the core of this thesis is but one aspect of equity in PES, which also includes procedural and distributional equity issues, such as access, decision-making, outcomes and legitimacy (Corbera et al., 2007a, Corbera et al., 2007b, McDermott et al., 2012). In the literature, PES is presented as a consultative process that involve for example, bargains between providers and buyers, and where the local farmers and communities are expected to play an important role in decision making. In practice, this is not always the case. The experiences documented from the two case studies analyzed showed that the implementation of these PES schemes did not always involve participatory consultation in many cases as reported by the majority of the landholders interviewed in the two study sites. It is critical to note the observation that the lack of consultation concerning the different aspects of PES such as rates, payment methods and other design issues did not necessarily discourage potential providers from enrolling and participating in these PES programs. This is clearly demonstrated in both the WLP and the OOC PES programs where despite the contention that they were not adequately consulted, landowners still enrolled and participated in these PES programs.

Moral hazards

The moral hazards in PES include leakages, perverse incentives, the problem of free-riders and strategic behavior in PES implementation (Salzman, 2005). Leakages (or spillages) can occur in a PES program when it leads to the displacement of threats to ecosystem conservation for example, when residents holding contracts convert or harvest from substitute ecosystems that would not have been exploited in the absence of contracts.

Leakage can undermine successful ecosystem service generation, to the extent that environmentally-damaging activities are merely displaced rather than reduced. Perverse incentives can occur because of the subsidy-like structure of many PES programs. This is pertinent in PES schemes where people are paid to avoid activities that would be illegal in nature, or when people are paid to undertake activities that they would have carried out in the absence of payments.

Lastly, the financial sustainability of PES in a dynamic global environment dominated by the fluidity of market forces and technological progress has also been question with the argument that in this context, PES may not guarantee natural ecosystem protection in perpetuity (McCauley, 2006).

Conclusion and the implications of study findings

As discussed in Chapter 2, there are four hypothesized relationships between biodiversity and poverty. The ideal situation regarding the desired outcomes for changes in biodiversity and poverty for PES intervention with the dual objectives of biodiversity conservation and poverty reduction is twofold. The first, is to avoid a *win-lose scenario,* where a decline in poverty is accompanied by a decline in biodiversity and/or a *lose-lose scenario*, where an increase in poverty is accompanied by a decline in biodiversity. The second would be to achieve a *win-win* scenario, where a decline in poverty is accompanied by an increase in biodiversity and/or a *win more-lose less,* scenario where a decline in poverty is accompanied by biodiversity conservation policies. Although Payment for Ecosystem Services (PES) programs are implemented in different socio-ecological contexts targeting different ecosystem services, this thesis is restricted to looking at direct payment PES for biodiversity conservation targeting wildlife conservation in the Kenyan ASALs.

Four broad conclusions emerge from this study which cover the following issues: the direct and indirect drivers of change affecting the interactions among biodiversity, ecosystem services, and human well-being (including poverty reduction) in Kenyan ASAL regions; the changes in the provision of ecosystem services in Kenyan ASAL areas and the potential of PES to increase the supply of habitat services to enable wildlife conservation in private and communal lands owned individually and collectively by pastoral communities; the potential and actual effects of PES on poverty conditions among pastoral communities and its implications on the dynamic pastoral livelihoods; and lastly, the financial sustainability of PES programs and its application in the ASAL areas *vis a vis* other conservation mechanisms.

The **first main conclusion** of this study is that significant loss of biodiversity (based on wildlife biomass as proxy) and major changes in selected provisioning, habitat and cultural and amenity ecosystem services of tourism have occurred in Kenyan ASAL areas between 1970s and mid-2000s. These changes are partly driven by climate change, land tenure policies and demographic change leading to changes in pastoral poverty conditions.

Based on the official statistics, this study established that the Kenyan ASALs recorded a high population growth rate between 1979 and 2009. The high population growth rate in the ASALs reflects a similar trend documented for dryland systems globally, which experienced the highest population growth rates in the 1990s of all the systems examined in the Millennium Ecosystem Assessment report.

Policy changes in land tenure based on the establishment of western notions of property rights to land, principally through privatization and sub-division of rangelands from large communally owned to small individually owned land units have been promoted across the ASAL regions in Africa (Lane, 1998). This process, which is predicated on the neo-classical economic (NCE) framework, is considered unsuitable for the ecological sustainability and resilience of rangelands. It has been documented in other studies that land subdivision in rangelands leads to a decrease in the physical scale over which livestock can range, and the social scale of ownership and management (Mwangi and Ostrom, 2009a).

Despite the social and ecological risks it presents, land privatization and subdivision in the study areas has facilitated the implementation of direct payment for biodiversity PES programs. Pastoral landowners can only be allowed to enroll and be paid from the PES schemes if they possess the land to be covenanted, and are able to present title deeds as a proof of ownership of land. In land based PES schemes such as the two cases reviewed in this thesis, a pre-requisite for PES to benefit the poor is to ensure equitable land distribution during the process of sub-division, and to allow participating landowners to also facilitate reciprocal access to pasture on PES enrolled land by non-participants during dry season and drought periods.

About 70% of the Group Ranches in Kenyan ASAL were sub-divided among individual owners between 1960 and 1990 (Republic of Kenya, 2012b). It is thus foreseen that if this trend does not change, the current demand for land privatization and sub-division by pastoral communities in Kenyan ASAL is likely to continue in the future. While the government policies have heretofore promoted rangeland privatization, this may change given that recent policy and legislative changes brought about by the 2010 Constitution have established new land policy, pieces of legislations, and institutions that may reverse this trend if they are effective. In the *Sessional Paper No. 3 of 2009 on the National Land Policy* for example, the Government of Kenya specifically recognized the special status of pastoral lands and the need to address issues affecting pastoral lands separately from the rest of the country. According to the Land Policy, the government seeks to secure pastoral livelihoods and tenure to land, by among other measures, the establishment of suitable methods for defining and registering land rights in pastoral areas while allowing pastoralists to maintain their unique land systems and livelihoods (Republic of Kenya, 2012a). If implemented, it can provide for the establishment of secure rights to land without necessarily privatizing land, by for example, the issuance of share certificate to legitimate registered members of group ranches in pastoral areas.

An issue of concern regarding privatization is the potential increase in land fragmentation with potential detrimental consequences for livestock mobility and wildlife migration (Galvin et al., 2008). As documented in this study, in the already sub-divided rangelands, PES particularly through direct payments provides a mechanism to keep rangelands open by stipulating land use regulations that prohibit practices that promote land fragmentation such as fencing and crop cultivation.

The establishment and rapid expansion of community conservancies (Chapter 3) also provides a mechanism to forestall future land sub-division, particularly if the benefit sharing arrangements adopted are equitable and transparent for member landholders. There is thus an opportunity for the national and county governments, NGOs, development partners and international financial institutions to support Conservancies as a way to forestall land subdivision and also to promote communal land practices that benefit land holders while helping to control land fragmentation by maintaining open rangelands.

The importance of climate change as a direct driver of change in ASALs cannot be overemphasized. In the short to medium term, the effects of climate change in ASALs will be manifested in terms of flood and drought occurrences, which present considerable risks to pastoral livelihoods. The focus of PES with regard to climate change has so far been heavily biased towards mitigation with little or no attention to climate change adaptation. This study has generated insights on the role of PES in climate change adaptation in ASALs, specifically in relation to drought risk coping strategy among participating pastoral families.

These insights are relevant to policy at both the national and global levels through the United Nations Framework Convention on Climate Change (UNFCCC) processes. For example, the role of PES in climate change adaptation in ASALs is not addressed in the Kenya Climate Change Response Strategy (Government of Kenya, 2010), and should therefore be incorporated in the subsequent climate change implementation plans. As discussed below, this role could be critical towards poverty prevention and addressing pastoral vulnerability to climate change by strengthening the adaptive capacity of pastoral PES participants.

The **second main conclusion** of this study is that PES can support wildlife conservation in ASALs but could also generate risks through spillovers and leakages it produces. While this study did not assess the impact of PES on biodiversity in its many dimension, it established that PES schemes assessed promoted the conservation of wildlife in two mains ways; by increasing the provision of wildlife habitat services in private and communal lands thereby supplementing formal state protected areas, and by helping to mitigate human-wildlife conflicts especially within the wildlife dispersal lands and corridors.

Concerning the provision of wildlife habitat services, the area of land protected or conserved for biodiversity is now considered as one of the indicators of national conservation progress. The fifth World Parks Congress in 2003 supported calls by conservation agencies for increasing protected area coverage, but also noted that this increase might come from the formal acceptance of the traditional forms of protected areas, including models with alternative governance structures (Langholz and Krug, 2004).

The implication here is that community conserved areas, co-managed areas, private parks and indigenous reserves, most of which are implemented through PES approaches, can be recognized as constituting protected areas (Carter et al., 2008). By supporting the protection of habitats in private and communal lands, PES can make a critical contribution towards meeting national and global targets. These include the UN Millennium Development Goals (MDGs) Goal 7 – Ensuring Environmental Sustainability – that includes protected area coverage as an indicator to measure progress, and Target 11 of the UN Convention on Biological Diversity (CBD) Strategic Plan for Biodiversity 2011-2020 which states that;

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

Two issues are however pertinent in the consideration of land under PES as protected areas. The first is the need to go beyond the quantity to the quality; it is not just the area of land that matters for wildlife, but much more important is the quality of the land and habitats therein. Here PES Conditionality can help to ensure that conservation results involving habitat quality are achieved. Second, it should be noted that, the setting aside of critical areas for wildlife through PES can also generate risks of spillages if not planned holistically based on the ecosystem approach. For example in the Olare Orok Conservancy, there are concerns that the displacement of settlements and livestock from the Conservancy could be contributing towards the increased illegal grazing inside the

Maasai Mara National Reserve which is a formal protected area , and also to overcrowding in the pastoral commons thereby undermining biodiversity conservation in these areas.

In relation to the role of PES in the mitigation of human-wildlife conflicts, this study has demonstrated using the WLP PES scheme, that PES can serve as a tool for mitigating livestock depredation by large. This finding strengthens the case for increased use of PES as a mechanism for promoting human-carnivore co-existence and poverty reduction in the developing countries (Dickman et al., 2011).

The **third main conclusion** of this study concerns the implications for PES on poverty. The evidence here is mixed because PES implementation generates both negative and positive outcomes on poverty, which affects both participants and non-participants differently. The conclusions derived for the four sequential questions on the links between PES and poverty outlined in Chapter 2 are as follows;

1. *To what extent do the poor people participate in PES schemes?*

It is generally assumed that household participation in PES schemes is a voluntary choice, but in the PES programs analyzed in this study, it is shown that the choice of whether a household is able to participate in a PES scheme or not also depends on other contextual factors beyond the household decision. These contextual factors include (a) funding availability, where limited funds would mean that a large number of willing households are excluded as is the case in the WLP. Here participation depends more on the decision of PES scheme managers rather than the households themselves; (b) the unseen pressures acting upon household decisions such as the increased cost of not participating and coercion. This is the case particularly where the PES scheme involves contiguous land holdings such as in the Olare Orok Conservancy. Here, if a landholding is located in the middle of other enrolled landholdings, the costs of not participating could be much higher for unwilling landowners because of the practical difficulties in access and the costs of monitoring livestock to avoid infractions on Conservancy land; and (c) lastly, poor households need for immediate cash income may also induce involuntary participation to enable them obtain money to meet short-terms needs. In this case, poor household may enroll or choose to participant but at the expense of long term wellbeing in terms of giving up critical pasturelands.

An analysis of the determinants of participation and intensity of participation which considered the WLP reveals that the PES scheme is not pro-poor except for poor households occupying areas of low or poor quality pasture in the Athi-Kaputie Plains. Otherwise the evidence shows that poor families with fewer assets (including land, livestock and cash income) have a lower probability of participation in PES compared to non-poor families. Furthermore, if poor families enroll, then they are likely to benefit less from PES compared to their non-poor counterparts.

There are potentially several reasons why the two PES schemes assessed are not pro-poor in terms of participation. The first concerns the eligibility requirement of proof of land ownership. This condition excludes poor landless families from participating, and affects particularly women who are traditionally excluded from land ownership and who may have customary access to family land but lack legal force to back it. The second concerns the initial land distribution during sub-division and prior to the establishment of the PES schemes. Inequitable distribution of land results in landless people, but also favors the local elites who are able to amass more land than the agreed average allocation per family and often in prime locations.

Corrective actions are necessary to make these PES programs pro-poor in terms of participation. For example, households and women without title deeds but with *de-facto* control of land by virtue of customary recognition should be allowed to enroll in the PES programs.

In addition, the following recommendations should be considered;

- Ensure that PES programs are designed to allow for the participation of the poor, particularly women. A practical way to implement this recommendation in both the WLP and the OOC PES schemes is for the PES managers to cease the use of title deeds as an eligibility requirement for PES enrolment. Here lessons can be learned from Costa Rica where the national PES programme which initially included a provision for title deeds, was revised to make the programme accessible to land users without title deeds (Pagiola, 2008).

- Pay more attention to gender inclusiveness in PES implementation. Here deliberate efforts are needed to ensure women are accommodated in PES schemes. Deliberate efforts should be made to include the participation of women, especially widows and unmarried women who have traditionally not been able to own land in pastoral areas.
- Studies are required to assess the gender impacts of PES implementation among pastoral communities as there is a major knowledge gap in this topic. One of the concerns relate to the fact that PES removes intra-household sharing of benefits related to land and natural resources and instead channels those benefits to the usually male household head.

2. *Does participation in PES make the poor sellers better-off?*

This study has established that participation generates both income and non-income effects for the households that have enrolled in the PES programs. In terms of the income effects, PES schemes generally make the participants, including the poor households better-off financially because it serves as a source of cash income diversification. More importantly, PES income is shown to be critical in periods of a severe drought such as in 2008-2009. During this drought, the PES share of household cash income increased substantially offsetting declines in the livestock share of household income due to the drought effect. This provides evidence that PES serves as a “safety-net” for participating pastoral families when faced with climatic shocks such as droughts that increase livestock mortality and diminish income from livestock assets. There is so far little documentation of this critical role of PES as a drought coping mechanism, and this study advances empirical evidence to support this observation.

Although this study has not established whether PES can lead to poverty reduction among participants (lift poor people out of poverty), it does provide some evidence that the poverty co-benefits of PES include both poverty alleviation (addressing some symptoms of poverty but not necessarily lifting poor households above poverty line) and poverty prevention (preventing people from falling into – or further into – poverty). Two recommendations suffice here;

- There is a need for greater consideration for the combined implementation of mechanisms for biodiversity conservation and poverty reduction, particularly PES and Conditional Cash Transfers (CCT) respectively as already suggested in the literature (Rodriguez et al., 2011b, Persson and Alpizar, 2013). The potential advantage of this approach is the possibility of widening the PES funding base, including the possibility to tap into poverty reduction funds on the strength of demonstrated evidence of the poverty reduction co-benefits of PES. This is already happening in the South Africa’s Working for Water (WfW) national PES programme which is funded from a national poverty reduction fund (Turpie et al., 2008).
- There is need for increased recognition among policy makers in government, international financial institutions and the climate change community in general, of the hitherto unrecognized but critical role of PES in climate change adaptation among pastoral communities. Further studies are needed to determine the extent to which PES programs can promote or undermine ecosystem based adaptation to climate change in the ASAL areas.

The non-income effects of PES among participants are mixed and can be positive or negative depending on the PES promoted land use regulations. In cases where the supported land use practices complements traditional pastoral livelihoods through integrated co-management of livestock and wildlife as is the case in the WLP, the non-income benefits are considered largely as positive to pastoral land owners. However, in cases where the PES supported land use practices do not fully complement traditional pastoral livelihoods for instance through seasonal restrictions of access to pasture as is the case in Olare Orok Conservancy, it can lead to negative non-income effects on pastoral landowners. It also implies that a higher rate of PES payment is required for participating households to be able to offset the opportunity costs of foregoing pasture grazing inside the Conservancy and to enable landowners to bear the trade-off involved. The tourism sector as “user” in wildlife based PES schemes has promoted land uses that exclude livestock grazing thereby negatively affecting pastoral households in terms of restricted access to traditional dry season pasture.

Three recommendations are critical in this situation;

- The implementation of PES in pastoral areas should be designed to support the seasonal mobility of herders and their livestock and to promote land uses that support integrated livestock-wildlife management rather than total exclusion of livestock.
- There is a need in PES programs to promote the integration of tourism and pastoralism. This approach can enable participating households to maintain flexibility in livelihoods. One practical way to achieve this is to educate tourists on the importance of livestock grazing to wildlife

3. *Do poor service buyers become better off from PES?*

All the buyers involved in the PES schemes assessed are institutions, and not individuals or households; hence this question was not examined in this thesis. This is an area that requires further research

4. *How are other non-participant poor affected by PES outcomes?*

This study has shown that PES implementation has affected non-participants at the study sites, which include the poor in a variety of ways. First, while PES has the effect of reducing income inequity among the participants, it has at the same time widened the income disparity between participants and non-participants, as observed in the Olare Orok Conservancy. Second, PES can support or undermine the adaptive capacity of both participants and non-participants with respect to opportunities for pastoral mobility. This depend on whether the PES promoted land use restrictions leads to open rangelands, and whether the rules allow for access by livestock herders to grazing lands.

Third, PES would be expected to have an effect on the land markets and rural labor, but the nature of these effects is yet to be established in the study areas. The “land diversion” PES programs, including the WLP and the OOC PES scheme serve to restrict diversification to crop production by pastoral participants. In the case of OOC, PES also limits non-participants reciprocal access to pasture potentially leading to conflicts because OOC members are allowed access to non-participants land and pastoral commons for livestock grazing. Consequently, non-participants indirectly bear the costs of PES implementation through crowding of the commons resulting from the limited grazing access to the Conservancy by both participants and non-participants.

Several recommendations are suggested;

- Although it may not be possible for a PES program to minimize the existing income inequality between participants and non-participants, it is recommended that ecosystem service users in PES program should consider the implementation of complementary initiatives to PES that also benefit non-participants. This is already happening in the OOC through the Olare Orok Conservancy Trust (OOCT) and should continue
- There is need for studies to assess the implication of PES implementation in the study areas on the land markets, rural labor and employment opportunities and in the management of the commons

The **last main conclusion** from this study is that both PES programs assessed lack financial sustainability. Based on the existing PES literature, differences are expected between “user financed” self-organized PES programs that are private sector led and “government-funded” PES programs, with the former expected to be more financially sustainable compared to the latter.

The WLP which is a “government-funded” PES program is severely constrained by lack of finances. The program has not been able to make upward adjustments to rates paid to participants since it was established in 2000. . Currently, all the institutions that made funding available to the WLP have had their priority shift to other landscapes (TNC for example), have conclude their grant arrangement (The World Bank/GEF) or are yet to make a decision whether to extend their funding support (KWS for example). Consequently, while the TWF as the implementer of the WLP is currently in the process of soliciting funds, there is uncertainly if this PES program will continue and for how long.

The OOC is a "user-funded" PES scheme and has so far not experienced financial challenges in terms of being unable to pay landowners enrolled in the program. The lack of financial sustainability in this program emerges from the volatility that exists in the tourism industry which is highly susceptible to political, economic and environmental shocks. It is further compounded by the unpredictability of tourism investors that can easily withdraw from joint ventures with local communities in the event that their businesses collapse or even in a situation of serious conflicts with land owners.

To ensure financial sustainability, the following recommendations are proposed;

- Both PES schemes in the WLP and the OOC can consider the establishment of a Conservation or Environmental Trust Fund (C/ETF) as a mechanism to ensure long term financial sustainability. as is already becoming common in other regions of the developing world. CTF is a central pool of ecosystem finance that is centrally managed by an entity that is independent from the institutions from which financing is generated. CTF may be managed as one of three different types of funds: *Endowments* invests principal capital in perpetuity and only investment income or interest on the endowment is spent; *Sinking funds* spend a portion of the principal investment along with the investment income; *Revolving funds* are maintained by earmarked revenue generated through taxes and fees among others (Global Canopy Programme (GCP), 2010). In addition to providing a stable source of funding, CTFs can also benefit the conservation community by promoting coordination among various stakeholders, including NGOs, governments, community groups and the private sector, by providing technical assistance in the design and implementation of conservation strategies, and by building local capacity for biodiversity conservation (WWF, 2009). CTFs can open up opportunities for additional grants from the private sector, philanthropic institutions and development grants.

- In the case of the WLP, financial sustainability can be improved in two other ways. The first option is to link the payments provided through the PES program to the user fees for Nairobi National Park. The rationale here is based on the fact that that the park is part of a wider ecosystem and its viability both as a wildlife refuge and as a tourist attraction is dependent on maintaining the wildlife dispersal areas in the AKP. This proposed approach is a form of Revenue Sharing (RS) scheme implemented through a PES model. The major difference in the proposed approach to the current funding generated from the KWS which is in charge of NNP is the fact that current KWS funding is provided as a time limited grant on an "*ad-hoc*" basis and is seen as a "favor" extended to the AKP landowners and can be terminated at any time. In contrast, the proposed model will be based on an explicit recognition that AKP landowners provide "habitat services" on their private land that is critical for the sustainability of wildlife in the entire ecosystem, and which attracts tourists to the NNP. These landowners are therefore entitled to part of the revenue generated from visitor entrance fee to NNP. As long as the park attracts visitors, so long will money be available to support the WLP thus ensuring funding sustainability. The second option is to explore the possibility of charging residents of Nairobi metropolis and the satellite towns that benefit from the ecosystem services within AKP and use this money generated to fund the WLP PES initiative. This recommendation has already been assessed and found to be economically feasible (Rodriguez et al. 2012).
- There is need to complement PES with other conservation mechanisms to create a portfolio of interventions rather than rely on PES as a single mechanism. For example, in the AKP, the WLP is complements the "*Predator Consolation Program*" that provides direct modest compensation to families that suffer livestock losses through predation. Other conservation mechanisms that can

complement PES include conservation easements, land purchases and the traditional integrated conservation and development projects (ICDPs).

Study Limitations and Retrospective Reflections

This study has some limitations that are highlighted here. While some of these were unavoidable, a few others are a result of the study design, focus and approach, and in retrospect, could have been minimised or addressed prior to and during the research process. These limitations include;

1. Data challenges **–** some of the data used especially the official population and poverty statistics for the ASAL regions are not very reliable. In addition, these are also dated, being more than 10 years. I was unable to obtain more recent data on poverty in the ASALs.
2. The study followed an ecological economics approach in the data collection and analysis as well as the interpretation of the results. This led to obvious gaps especially in terms of the study being largely ahistorical (failing to capture the historical processes), apolitical (not taking sufficient cognisance of political dynamics especially concerning land sub-division, distribution and allocation processes), and not sufficiently integrating the landscape perspective in the analysis of the PES programs. Retrospectively, these limitations would have been addressed to an extent by combining the ecological economics approach adopted together with the political ecology and political economy frameworks, which would have provided the conceptual framework to address the historical, political and landscape perspectives.
3. The study design and analysis could have been structured to include an in-depth assessment of costs at more or less the same level as are the benefits. In addition, it also could have been designed to include the voices that are currently not represented in the study. As it were, in terms of the case studies assessed, the study focussed only on the landowners, including those that are participating in the PES schemes and those that are not participating, the intermediary institutions that are facilitating the PES schemes, and the institutions providing the financing for PES implementation. The study would have been enriched if

the voices of other more vulnerable groups were also included, through interviews. These include the elderly, the women, the youth and the landless. Because of these missing voices, little attention is paid to their perspectives overall. As a result, the following issues, which are not well addressed in the thesis, would be informative in any future research on the PES schemes in the two study areas;

a. A detailed cost-benefit analyses (CBA) of the PES schemes to provide insights on the allocation of costs and benefits among the different stakeholder groups. While the thesis has looked in-depth into the income gains of households and the revenue flows from PES, it would have benefited also to assess the revenue flows of the tourism entrepreneurs to provide a balanced perspective and analysis of the relative benefits and costs for both groups.
b. The equity implications of PES in terms of gender dynamics, intra-household resource use and allocation, intergenerational equity. A scenario analysis would have brought forth insights on the perspectives of the elderly and the youth regarding land management and PES schemes to provide better insights on the sustainability of the schemes from the land owner's perspectives.
c. The politics and dynamics of land sub-division process prior to and during the establishment of PES schemes to determine who benefits, who loses, how are the land allocation process conducted, whose voice counts. Herein the motives of the investors in PES schemes are critical but not wholly brought forward. More importantly as well is the source of capital for PES investments and the message being used by the investors to justify the investments in PES schemes. The risk of land dispossession through land sales and long term conservancy contracts may be imminent for the land owners given the experiences in other parts of the Maasailand.

REFERENCES CITED

ABEYASEKERA, S. (2005) Multivariate methods for index construction. *Household sample surveys in Developing and Transition Countries.* New York, United Nations.

ABOUD, A. A., KISOYAN, P., SAID, M., DE LEEUW, J., NOTENBAERT, A., DAVIES, J., MANZANO, P., OMONDI, S. & WAITHAKA, M. (2012a) Payment for wildlife conservation in the Maasai Mara Ecosystem. IN GITAU, J. (Ed.) *ASARECA ABCD Series Policy Brief no.2.* ASARECA (Association for Strengthening Agricultural Research in East & Central Africa).

ABOUD, A. A., KISOYAN, P., SAID, M., NOTENBAERT, A., DE LEEUW, J., GITAU, J., MANZANO, P., DAVIES, J., ROBA, G., OMONDI, S. O. & ODHIAMBO, M. (2012b) Natural Resource Management and Biodiversity Conservation in the Drylands of Eastern and Central Africa. Entebbe, Uganda, ASARECA (Association for Strengthening Agricultural Research in Eastern and Central Africa).

ACC (2005) Will Kitengela die? Nairobi, Kenya, African Conservation Center.

ACHESON, J. M. (2006) Institutional Failure in Resource Management. *Annual Review of Anthropology,* 35**,** 117-134.

ADAMS, W. M., AVELING, R., BROCKINGTON, D., DICKSON, B., ELLIOTT, J., HUTTON, J., ROE, D., VIRA, B. & WOLMER, W. (2004) Biodiversity conservation and the eradication of poverty. *Science,* 306**,** 1146-1149.

ADAMS, W. M. & HUTTON, J. (2007) People, Parks and Poverty: Political Ecology and Biodiversity Conservation. *Conservation & Society,* 5**,** 147-183.

AGRAWAL, A. (2008) The Role of Local Institutions in Adaptation to Climate Change. *IFRI Working Paper.* IFRI (International Forestry Resources and Institutions) Program.

AGRAWAL, A. & REDFORD, K. H. (2006) Poverty, Development, And Biodiversity Conservation: Shooting in the Dark. *WCS Working Paper Series.* Wildlife Conservation Society (WCS).

AKAMA, J. S. (1999) Marginalization of the Maasai in Kenya. *Annals of Tourism Research,* 26**,** 716-718.

AKAMA, J. S. (2002) The role of government in the development of tourism in Kenya. *International Journal of Tourism Research,* 4**,** 1-13.

ALERS, M., BOVARNICK, A., BOYLE, T., MACKINNON, K. & SOBREVILA, C. (2007) Reducing Threats to Protected Areas: Lessons from the Field. Washington D.C. and New York, USA, World Bank and UNDP (United Nations Development Programme).

ALKIRE, S. & FOSTER, J. (2011) Counting and multidimensional poverty measurement. *Journal of Public Economics,* 95**,** 476-487.

ANDAM, K. S., FERRARO, P. J., SIMS, K. R. E., HEALY, A. & HOLLAND, M. B. (2010) Protected areas reduced poverty in Costa Rica and Thailand. *Proceedings of the National Academy of Sciences,* 107**,** 9996-10001.

ANDERSON, D. & BROCH-DUE, V. (Eds.) (1999) *The poor are not us: poverty and pastoralism in East Africa,* Oxford, James Currey.

ARAAR, A. & DUCLOS, J.-Y. (2009) DASP: Distributive Analysis Stata Package. *User Manual (DASP Ver 2.1).* Universite Laval, PEP, CIRPEE and World Bank.

ARMSWORTH, P. R., DAILY, G. C., KAREIVA, P. & SANCHIRICO, J. N. (2006) Land market feedbacks can undermine biodiversity conservation. *Proceedings of the National Academy of Sciences,* 103**,** 5403-5408.

ARSEL, M. & BÜSCHER, B. (2012) Nature™ Inc.: Changes and Continuities in Neoliberal Conservation and Market-based Environmental Policy. *Development and Change,* 43**,** 53-78.

ASHLEY, C. (1995) Tourism, Communities and the Potential Impacts on Local Incomes and Conservation. *Research Discussion Paper No.8.* Windhoek, Namibia, Directorate of Environmental Affairs, Ministry of Environment and Tourism, Government of Namibia.

ASHLEY, C. & ELLIOT, J. (2003) "Just Wildlife?" or a Source of Local Development? *Natural Resources Perspective*.

ASHLEY, C. & ROE, D. (1998) Enhancing Community involvement in wildlife tourism: Issues and challenges. *IIED Widlife and Development Series no 11.* London, UK, International Institute for Environment and Development (IIED).

ASQUITH, N. M., VARGAS, M. T. & WUNDER, S. (2008) Selling two environmental services: In-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. *Ecological Economics,* 65**,** 675-684.

AWF (2009) Wildlife and Communities Gain New Lease on Land. Nairobi, AWF (African Wildlife Foundation).

BALVANERA, P., DAILY, G. C., EHRLICH, P. R., RICKETTS, T. H., BAILEY, S.-A., KARK, S., KREMEN, C. & PEREIRA, H. (2001) Conserving Biodiversity and Ecosystem Services. *Science,* 291**,** 2047-.

BARBIER, E. (2011) The policy challenges for green economy and sustainable economic development. *Natural Resources Forum,* 35**,** 233-245.

BARNES, J., MACGREGOR, J. & ALBERTS, M. (2012) Expected climate change impacts on land and natural resource use in Namibia: exploring economically efficient responses. *Pastoralism: Research, Policy and Practice,* 2**,** 22.

BARRETT, C. B. (2005) Rural poverty dynamics: development policy implications. *Agricultural Economics,* 32**,** 45-60.

BARRETT, C. B., BRANDON, K., GIBSON, C. & GJERTSEN, H. (2001) Conserving Tropical Biodiversity amid Weak Institutions. *Bioscience,* 51**,** 497-501.

BARRETT, C. B., CHABARI, F., BAILEY, D., LITTLE, P. D. & COPPOCK, D. L. (2003) Livestock Pricing in the Northern Kenyan Rangelands. *Journal of African Economies,* 12**,** 127-155.

BARRETT, C. B., TRAVIS, A. J. & DASGUPTA, P. (2011) On biodiversity conservation and poverty traps. *Proceedings of the National Academy of Sciences,* 108**,** 13907-13912.

BARROW, E. & MOGAKA, H. (2007) Kenya's Drylands: Wastelands or an Undervalued National Economic Resource. IUCN (The World Conservation Union).

BEHNKE, R. (1994) Natural Resource Management in Pastoral Africa. *Development Policy Review,* 12**,** 5-28.

BEHNKE, R. (2008) The Drivers of Fragmentation in Arid and Semi-Arid Landscapes. IN GALVIN, K. A., REID, R. S., BEHNKE, R. & HOBBS, N. T. (Eds.) *Fragmentation in Semi-Arid and Arid Landscapes: Consequences for Human and Natural Systems.* Dordrecht, The Netherlands, Springer.

BEHNKE, R. & MUTHAMI, D. (2011) The Contribution of Livestock to the Kenyan Economy. *IGAD Livestock Policy Initiative (LPI) Working Paper No. 03-11.* Inter-Governmental Authority on Development (IGAD).

BENNETT, M. T. (2008) China's sloping land conversion program: Institutional innovation or business as usual? *Ecological Economics,* 65**,** 699-711.

BERKES, F. (2004) Rethinking Community-based Conservation. *Conservation Biology,* 18**,** 621-630.

BHOLA, N., OGUTU, J. O., SAID, M. Y., PIEPHO, H. P. & OLFF, H. (2012) The distribution of large herbivore hotspots in relation to environmental and anthropogenic correlates in the Mara region of Kenya. *Journal of Animal Ecology*.

BIGGS, R., BOHENSKY, E., DESANKER, P. V., FABRICIUS, C., LYNAM, T., MISSELHORN, A. A., MUSVOTO, C., MUTALE, M., REYERS, B., SCHOLES, R. J., SHIKONGO, S. & VAN JAARSVELD, A. S. (2004) Nature supporting people: the Southern Africa Millennium Assessment. Pretoria, Council for Scientific & Industrial Research (CSIR).

BIRCH, I. & GRAHN, R. (2007) Pastoralism - Managing Multiple Stressors And the Threat of Climate Variability and Change. *Human Development Report Office Occassional Paper 2007/45.* UNDP (United Nations Development Programme).

BIRNER, R. & OSANO, P. (2012) Community conservancies and payments for wildlife conservation (PWC) as a coping strategy under different conservancy institutional arrangements. *Workshop on "Enabling Livestock Based Economies in Kenya to Adapt to Climate Change: A Review of PES from Wildlife Tourism As A Climate Change Adaptation Option", February 15, 2012, ILRI, Nairobi, Kenya. Available Online: http://www.slideshare.net/ILRI/community-conservancies-and-payments-for-wildlife-conservation-pwc-as-a-coping-strategy-under-different-conservancy-institutional-arrangements.* ILRI, Nairobi, Kenya.

BISHOP, J., KAPILA, S., HICKS, F., MITCHELL, P. & VORHIES, F. (2008) Building biodiversity business. London, UK and Gland, Switzerland, Shell International Limited and the International Union for Conservation of Nature (IUCN).

BOJO, J. (1996) The Economics of Wildlife: Case Studies from Ghana, Kenya, Namibia and Zimbabwe. Washington, D.C., World Bank.

BOND, I., WAAGE, S. & RUHWEZA, A. (2008) Payments for Ecosystem Services in East and Southern Africa: Assessing prospects & pathways forward. The Katoomba Group.

BOSIRE, C., KARIUKI, J. & OSANO, P. (2012) Enabling Livestock Based Economies in Kenya to Adapt to Climate Change: A review of PES from wildlife tourism as a climate change adaptation option - A Workshop Report. Nairobi, Kenya, ILRI/University of Hohenhiem/ZALF.

BOYD, J. & BANZHAF, S. (2007) What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics,* 63**,** 616-626.

BRAAT, L. C. & DE GROOT, R. (2012) The ecosystem services agenda:bridging the worlds of natural science and economics, conservation and development, and public and private policy. *Ecosystem Services,* 1**,** 4-15.

BRANDON, K., REDFORD, K. H. & SANDERSON, S. E. (Eds.) (1998) *Parks in Peril: People, Politics and Protected Areas,* Washington DC, The Nature Conservancy/Island Press.

BROMLEY, D. W. (1991) *Environment and Economy: Property Rights and Public Policy,* Cambridge, MA, Blackwell Publishers.

BROOKS, T. M., MITTERMEIER, R. A., MITTERMEIER, C. G., DA FONSECA, G. A. B., RYLANDS, A. B., KONSTANT, W. R., FLICK, P., PILGRIM, J., OLDFIELD, S., MAGIN, G. & HILTON-TAYLOR, C. (2002) Habitat Loss and Extinction in the Hotspots of Biodiversity. *Conservation Biology,* 16**,** 909-923.

BROWN, G. P. (2008) *The Commonwealth of Life: Economics for a Flourishing Earth,* Black Rose Books.

BRUNER, A. G., GULLISON, R. E. & BALMFORD, A. (2004) Financial costs and shortfalls of managing and expanding protected-area systems in developing countries. *Bioscience,* 54**,** 1119.

BRUNER, A. G., GULLISON, R. E., RICE, R. E. & DA FONSECA, G. A. B. (2001) Effectiveness of parks in protecting tropical biodiversity. *Science,* 291**,** 125-128.

BULTE, E. H., BOONE, R. B., STRINGER, R. & THORNTON, P. K. (2008a) Elephants or onions? Paying for nature in Amboseli, Kenya. *Environment and Development Economics,* 13**,** 395-414.
BULTE, E. H., LIPPER, L., STRINGER, R. & ZILBERMAN, D. (2008b) Payments for ecosystem services and poverty reduction: concepts, issues, and empirical perspectives. *Environment and Development Economics,* 13**,** 245-254.
BURNHAM, K. P. & ANDERSON, D. R. (2002) *Model selection and multimodel inference: a practical information-theoritic approach,* New York, USA, Springer-Verlag.
BUTCHART, S. H. M., WALPOLE, M., COLLEN, B., VAN STRIEN, A., SCHARLEMANN, J. P. W., ALMOND, R. E. A., BAILLIE, J. E. M., BOMHARD, B., BROWN, C., BRUNO, J., CARPENTER, K. E., CARR, G. M., CHANSON, J., CHENERY, A. M., CSIRKE, J., DAVIDSON, N. C., DENTENER, F., FOSTER, M., GALLI, A., GALLOWAY, J. N., GENOVESI, P., GREGORY, R. D., HOCKINGS, M., KAPOS, V., LAMARQUE, J.-F., LEVERINGTON, F., LOH, J., MCGEOCH, M. A., MCRAE, L., MINASYAN, A., MORCILLO, M. H., OLDFIELD, T. E. E., PAULY, D., QUADER, S., REVENGA, C., SAUER, J. R., SKOLNIK, B., SPEAR, D., STANWELL-SMITH, D., STUART, S. N., SYMES, A., TIERNEY, M., TYRRELL, T. D., VIA, J.-C. & WATSON, R. (2010) Global Biodiversity: Indicators of Recent Declines. *Science,* 328**,** 1164-1168.
CARO, T. (2008) Decline of large mammals in the Katavi-Rukwa ecosystem of western Tanzania. *African Zoology,* 43**,** 99-116.
CARO, T. & SCHOLTE, P. (2007) When protection falters. *African Journal of Ecology,* 45**,** 233-235.
CARTER, E., ADAMS, W. M. & HUTTON, J. (2008) Private protected areas: management regimes, tenure arrangements and protected area categorization in East Africa. *Oryx,* 42**,** 177-186.
CASH, D. W. & MOSER, S. C. (2000) Linking global and local scales: designing dynamic assessment and management processes. *Global Environ. Change,* 10**,** 109.
CATLEY, A., LIND, J. & SCOONES, I. (Eds.) (2013) *Pastoralism and Development in Africa: Dynamic Change at the Margins,* London and New York, Earthscan and Routledge.
CDC/IISD/SAFERWORLD (2009) Climate change and conflict: Lessons from community conservancies in northern Kenya. CDC (Conservation Development Center), IISD (International Institute for Sustainable Development) and Saferworld.
CENTRAL BUREAU OF STATISTICS (2003) Geographic Dimensions of Well-Being in Kenya: Where are the poor?: From Districts to Locations (Volume 1). Nairobi, Central Bureau of Statistics (CBS), Ministry of Planning and National Development.
CENTRAL BUREAU OF STATISTICS (2005) Geographic Dimensions of Well-Being in Kenya: Who and Where are the Poor? A Constituency Level Profile. Vol. II. Nairobi, Central Bureau of Statistics, Ministry of Planning and National Development.
CESPA (2008) Situational Analysis of Ecosystem Services and Poverty Alleviation in arid and semi-arid Africa. CESPA (Consortium of Ecosystem Services and Poverty Alleviation in arid and semi-arid Africa).
CHEN, S. & RAVALLION, M. (2007) Absolute poverty measures for the developing world, 1981-2004. *Proceedings of the National Academy of Sciences,* 104**,** 16757-16762.
CHILD, B. (1996) The practice and principles of community-based wildlife management in Zimbabwe: The CAMPFIRE programme. *Biodiversity and Conservation,* 5**,** 369-398.
CHRISTIE, I. T. & CROMPTON, D., E. (2001) Tourism in Africa. *World Bank Working Paper Series.* Washington, D.C., The World Bank.

CLEMENTS, T., JOHN, A., NIELSEN, K., AN, D., TAN, S. & MILNER-GULLAND, E. J. (2009) Payments for biodiversity conservation in the context of weak institutions: Comparison of three programs from Cambodia. *Ecological Economics,* 69**,** 1283-1291.

COLLIER, P. (2007) Poverty reduction in Africa. *Proceedings of the National Academy of Sciences,* 104**,** 16763-16768.

COMMISSION ON REVENUE ALLOCATION (CRA) (2011) Kenya: County Fact Sheets. Nairobi, Commission on Revenue Allocation (CRA). Online www.crakenya.org (Accessed on May 10, 2012).

CORBERA, E., BROWN, K. & ADGER, W. N. (2007a) The Equity and Legitimacy of Markets for Ecosystem Services. *Development and Change,* 38**,** 587-613.

CORBERA, E., KOSOY, N. & MARTÍNEZ TUNA, M. (2007b) Equity implications of marketing ecosystem services in protected areas and rural communities: Case studies from Meso-America. *Global Environmental Change,* 17**,** 365-380.

CORBERA, E., SOBERANIS, C. G. & BROWN, K. (2009) Institutional dimensions of Payments for Ecosystem Services: An analysis of Mexico's carbon forestry programme. *Ecological Economics,* 68**,** 743-761.

COSTANZA, R., D'ARGE, R., DE GROOT, R., FARBER, S., GRASSO, M., HANNON, B., LIMBURG, K., NAEEM, S., O'NEILL, R. V., PARUELO, J., RASKIN, R. G., SUTTON, P. & VAN DEN BELT, M. (1997) The value of the world's ecosystem services and natural capital. *Nature,* 387**,** 253-260.

COUDOUEL, A., HENTSCHEL, S. J. & WODON, T. Q. (2002) Poverty Measurement and Analysis. IN KLUGMAN, J. (Ed.) *A Sourcebook for Poverty Reduction Strategies.* Washington DC, The World Bank.

COUPE, S., LEWIS, V., OGUTU, Z. A. & WATSON, C. (2002) Living with Wildlife: Sustainable Livelihood for Park-adjacent Communities in Kenya. IN SCOTT, A. (Ed.) *ITDG Working Paper Series.* London, Intermediate Technology Development Group (ITDG).

COURTNEY, C. H. A. (2009) An Investigation into the Extent to which Activities in the Olare Orok Conservancy, Narok District, Kenya, Conform to "Ecotourism". Masters Thesis, University of Edinburgh, UK.

CRAIGIE, I. D., BAILLIE, J. E. M., BALMFORD, A., CARBONE, C., COLLEN, B., GREEN, R. E. & HUTTON, J. M. (2010) Large mammal population declines in Africa's protected areas. *Biological Conservation,* 143**,** 2221-2228.

CURTIN, C. & WESTERN, D. (2008) Grasslands, people, and conservation: Over-the-horizon learning exchanges between African and American pastoralists. *Conservation Biology,* 22**,** 870-877.

DAILY, G. C. (Ed.) (1997) *Nature's Services: Societal Dependence on Natural Ecosystems,* Washington DC, Island Press.

DAVIES, J. (2008) Turning the tide: Enabling sustainable development for Africa's mobile pastoralists. *Natural Resources Forum,* 32**,** 175-184.

DE GROOT, K. (2011) Payments for environmental services (PES) from tourism - arealistic incentive to improve local livelihoods and sustain forest landscapes in Vietnam's northern highlands? , Masters Thesis, Wageningen University, The Netherlands.

DEATON, A. (2003) 'How to Monitor Poverty for the Millennium Development Goals'. *Journal of Human Development,* 4**,** 353-378.

DEFRIES, R., HANSEN, A., TURNER, B. L., REID, R. S. & LIU, J. (2007) Land Use Change Around Protected Areas: Management to Balance Human Needs and Ecological Function. *Ecological Applications,* 17**,** 1031-1038.

DEROSE, L., MESSER, E. & MILLMAN, S. (1998) Who's hungry? And how do we know? Food shortage, poverty, and deprivation. Tokyo, United Nations University Press.

DEVEAU, V. & MARSHALL, I. M. (2008) How Beneficial is Tourism? An Analysis of the Economic Impact of Tourism in Ill N'gwesi, Kenya. *American Agricultural Economics Association Annual Meeting, July 27-29, 2008.* Orlando, Florida, FL.

DFID (1999) Sustainable Tourism and Poverty Elimination Study. A Report to the Department of International Development. Delloite & Touche/International Institute of Environment and Development (IIED)/Overseas Development Institute (ODI).

DFID (2002) Wildlife and Poverty Study. London, UK Department for International Development.

DICKMAN, A. J., MACDONALD, E. A. & MACDONALD, D. W. (2011) A review of financial instruments to pay for predator conservation and encourage human-carnivore coexistence. *Proceedings of the National Academy of Sciences,* 108**,** 13937-13944.

DIEKE, P. U. C. (1991) Policies for tourism development in Kenya. *Annals of Tourism Research,* 18**,** 269-294.

DOUGILL, A. J., STRINGER, L. C., LEVENTON, J., RIDDELL, M., RUEFF, H., SPRACKLEN, D. V. & BUTT, E. (2012) Lessons from community-based payment for ecosystem service schemes: from forests to rangelands. *Philosophical Transactions of the Royal Society B: Biological Sciences,* 367**,** 3178-3190.

DU TOIT, J. T., COCK, R. & DEUTSCH, J. C. (2010) *Wild Rangelands: Conserving Wildlife While Maintaining Livestock in Semi-Arid Ecosystems*, Wiley-Blackwell.

DURAIAPPAH, A. K. (1998) Poverty and environmental degradation: A review and analysis of the nexus. *World Development,* 26**,** 2169-2179.

DURAIAPPAH, A. K. & MARLENE, R. (2007) Poverty and Ecosystems: Prototype assessment and reporting method - Kenya case study. Manitoba, Canada, International Institute for Sustainable Development (IISD).

DURNING, A. B. (1989) Poverty and Environment: Reversing the Downward Spiral. Washington, D.C., World Watch Institute.

DUTTILLY-DIANE, C., MCCARTHY, N., FRANCIS., T., BRUGGEMAN, A., TIEDEMANN, J., STREET, K. & SERRA, G. (2007) Could Payments for Environmental Services Improve Rangeland Management in Central Asia, West Asia and North Africa? *CAPRI Working Papers.* Washington D.C., International Food Policy Research Institute (IFPRI).

EARNSHAW, A. & EMERTON, L. (2000) The Economics of Wildlife Tourism: Theory and Reality for Landholders in Africa. IN PRINS, H. H. T., GROOTENHUIS, J. G. & DOLAN, T. T. (Eds.) *Wildlife Conservation by Sustainable Use.* Kluwer Academic Publishers.

EKBOM, A. & BOJO, J. (1999) Poverty and Environment: Evidence of Links and Integration in the Country Assistance Strategy Process. *World Bank Africa Region. Discussion Paper No. 4.* Washington, D.C., World Bank.

EMERTON, L. (1999) The Nature of Benefits and the Benefits of Nature: Why Wildlife Conservation Has Not Economically Benefitted Communities in Africa. Manchester, UK, Institute for Development Policy and Management (IDPM), University of Manchester.

EMI (2010) Africa Travel and Tourism: Building a Legacy for the Future. EuroMonitor International (EMI).

ENGEL, S., PAGIOLA, S. & WUNDER, S. (2008) Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics,* 65**,** 663-674.

ENGEL, S. & PALMER, C. (2008) Payments for environmental services as an alternative to logging under weak property rights: The case of Indonesia. *Ecological Economics,* 65**,** 799-809.

ERICKSEN, P., DE LEEUW, J., SAID, M., SILVESTRI, S. & ZAIBET, L. (2012) Mapping ecosystem services in the Ewaso Ng'iro catchment. *International Journal of Biodiversity Science, Ecosystem Services & Management,* 8**,** 122-134.

ERICKSEN, P., DE LEEUW, J., THORNTON, P. K., SAID, M., HERRERO, M. & NOTENBAERT, A. (2013) Climate change in Sub-Saharan Africa: what consequences for pastoralism? IN CATLEY, A., LIND, J. & SCOONES, I. (Eds.) *Pastoralism and Development in Africa: Dynamic Change at the Margins.* London and New York, Routledge.

ERIKSEN, S. H. & LIND, J. (2009) Adaptation as a Political Process: Adjusting to Drought and Conflict in Kenya's Drylands. *Environmental Management*, 43**,** 817-835.

ERIKSEN, S. H. & O'BRIEN, K. (2007) Vulnerability, poverty and the need for sustainable adaptation measures. *Climate Policy,* 7**,** 337-352.

ERVIN, J., MULONGOY, K. J., LAWRENCE, K., GAME, E., SHEPPARD, D., BRIDGEWATER, P., BENNET, G., GIDDA, S. B. & BOSS, P. (2010) Making Protected Areas Relevant: A guide to integrating protected areas into wider landscapes, seascapes and sectoral plans and strategies. *CBD Technical Series No. 44.* Montreal, Canada, Secretariat of the Convention on Biological Diversity.

FAIRHEAD, J., LEACH, M. & SCOONES, I. (2012) Green Grabbing: a new appropriation of nature? *Journal of Peasant Studies,* 39**,** 237-261.

FAO (2005a) AFRICOVER - Spatially aggregated multipurpose Landcover database. Rome, Italy, FAO (The Food and Agriculture Organisation of the United Nations).

FAO (2005b) KENYA Livestock Sector Brief. Rome, FAO (Food and Agriculture Organisation of the United Nations).

FAO (2007) The State of Food and Agriculture: Paying Farmers for Environmental Services. *FAO Agricultural Series* Rome, Food & Agricultural Organisation of the United Nations.

FAO (2011) Payments for Ecosystem Service and Food Security. Rome: Italy, FAO (Food and Agriculture Organisation of the United Nations).

FARLEY, J. & COSTANZA, R. (2010) Payments for ecosystem services: From local to global. *Ecological Economics,* 69**,** 2060-2068.

FERRARO, P. J. & KISS, A. (2002) ECOLOGY: Direct Payments to Conserve Biodiversity. *Science,* 298**,** 1718-1719.

FERRARO, P. J. & PATTANAYAK, S. K. (2006) Money for Nothing? A Call for Empirical Evaluation of Biodiversity Conservation Investments. *PLoS Biology,* 4**,** e105.

FEWSNET (2010) La Nina and Food Security in East Africa. Washington, D.C., FEWSNET (Famine Early Warning System in East Africa) and USAID.

FISHER, B., TURNER, K., ZYLSTRA, M., BROUWER, R., GROOT, R. D., FARBER, S., FERRARO, P. J., GREEN, R., HADLEY, D., HARLOW, J., JEFFERISS, P., KIRKBY, C., MORLING, P., MOWATT, S., NAIDOO, R., PAAVOLA, J., STRASSBURG, B., YU, D. & BALMFORD, A. (2008) ECOSYSTEM SERVICES AND ECONOMIC THEORY: INTEGRATION FOR POLICY-RELEVANT RESEARCH. *Ecological Applications,* 18**,** 2050-2067.

FISHER, B., TURNER, R. K. & MORLING, P. (2009) Defining and classifying ecosystem services for decision making. *Ecological Economics,* 68**,** 643-653.

FISHER, J. A., PATENAUDE, G., MEIR, P., NIGHTINGALE, A. J., ROUNSEVELL, M. D. A., WILLIAMS, M. & WOODHOUSE, I. H. (In press) Strengthening conceptual foundations: Analysing frameworks for ecosystem services and poverty alleviation research. *Global Environmental Change*.

FONNAP (2011) The 2011 Wet Season Game Count: Findings and Implication for FoNNAP Strategy. *Newsletter of the Friends of Nairobi National Park (FoNNAP) - August, 2011*. Nairobi, Kenya, Friends of Nairobi National Park.

FRANCO-MAASS, S., NAVA-BERNAL, G., ENDARA-AGRAMONT, A. & GONZALEZ-ESQUIVEL, C. (2008) Payments for environmental services: An alternative for sustainable rural development? The case of a National Park in the Central Highlands of Mexico. *Mountain Research and Development,* 28**,** 23-25.

FRASER, D. G. E., DOUGILL, J. A., HUBACEK, K., QUINN, H. C., SENDZIMIR, J. & TERMANSEN, M. (2011) Assessing Vulnerability to Climate Change in Dryland Livelihood Systems: Conceptual Challenges and Interdisciplinary Solutions. *Ecology and Society,* 16**,** 3.

FRATKIN, E. (1997) Pastoralism: Governance and Development Issues. *Annual Review of Anthropology,* 26**,** 235-261.

FROST, P. G. H. & BOND, I. (2008) The CAMPFIRE programme in Zimbabwe: Payments for wildlife services. *Ecological Economics,* 65**,** 776-787.

FRYXELL, J. (1995) Aggregation and migration by grazing ungulates in relation to resources and predators. IN SINCLAIR, A. R. E. & ARCESE, P. (Eds.) *Serengeti II. Dynamics, management and conservation of an ecosystem.* Chicago, University of Chicago Press.

FUNK, C., EILERTS, G. & DAVENPORT, F. (2010) A climate trend analysis for Kenya - August 2010. Farming Early Warning Systems Network (FEWSNET).

FYNN, R. W. S. & BONYONGO, M. C. (2010) Functional conservation areas and the future of Africa's wildlife. *African Journal of Ecology,* 49**,** 175-188.

GALATY, J. G. (1981) Land and Livestock among Kenyan Masai: Symbolic Perspectives on Pastoral Exchange, Change and Inequality. IN GALATY, J. G. & SALZMAN, P. C. (Eds.) *Change and Development in Nomadic and Pastoral Societies.* Leiden, E. J. Brill.

GALATY, J. G. (1992) "The Land is Yours": social and economic factors in the privatization, sub-division and sale of Maasai ranches. *Nomadic Peoples,* 30**,** 26.

GALATY, J. G. (1994a) Ha(l)ving land in common: the subdivision of Maasai group ranches in Kenya. *Nomadic Peoples,* 34/35**,** 109-122.

GALATY, J. G. (1994b) The vanishing African herdsman. *UNESCO Courier,* 47**,** 28.

GALATY, J. G. (1999) Grounding Pastoralists: Law, Politics, and Dispossesion in East Africa. *Nomadic Peoples,* 3**,** 56-73.

GALVIN, K. A. (2009) Transitions: pastoralists living with change. *Annual Review of Anthropology,* 38**,** 185 - 198.

GALVIN, K. A., REID, R. S., BEHNKE, R. & HOBBS, N. T. (Eds.) (2008) *Fragmentation in Semi-Arid and Arid landscapes: consequences for Human and Natural Systems,* Dordrecht, Springer.

GALVIN, K. A., THORNTON, P. K., BOONE, R. B. & SUNDERLAND, J. (2004) Climate variability and impacts on east African livestock herders: the Maasai of Ngorongoro Conservation Area, Tanzania. *African Journal of Range & Forage Science,* 21**,** 183-189.

GALVIN, M. & HALLER, T. (Eds.) (2008) *People, Protected Areas and Global Change: Participatory Conservation in Latin America, Asia, Africa and Europe,* Bern, University of Bern.

GAUVIN, C., UCHIDA, E., ROZELLE, S., XU, J. & ZHAN, J. (2009) Cost-Effectiveness of Payments for Ecosystem Services with Dual Goals of Environment and Poverty Alleviation. *Environmental Management,* 45**,** 488-501.

GEF (2008) Wildlife Conservation Leasing Demonstration Project: Project Document for CEO Approval (Revised). Global Environment Facility (GEF) Project Database.

GEF (2010) Payment for Ecosystem Services. Washington D.C., GEF (The Global Environment Facility).
GIBBONS, J. M., NICHOLSON, E., MILNER-GULLAND, E. J. & JONES, J. P. G. Should payments for biodiversity conservation be based on action or results? *Journal of Applied Ecology,* 48**,** 1218-1226.
GICHOHI, W. H. (1996) The ecology of a truncated ecosystem:The Athi-Kapiti Plains. PhD Thesis, University of Leicester.
GICHOHI, W. H. (2003) Direct payments as a mechanism for conserving important wildlife corridor links between Nairobi National Park and its wider ecosystem: The Wildlife Conservation Lease Program. *Vth World Parks Congress, September, 2003, Durban, South Africa.* Durban, South Africa.
GLOBAL CANOPY PROGRAMME (GCP) (2010) The Little Biodiversity Finance Book: A guide to proactive investment in natural capital (PINC). Oxford, UK, Global Canopy Programme (GCP).
GOLDMAN, M. J. (2011) Strangers in their own land: Maasai and wildlife conservation in Northern Tanzania. *Conservation & Society,* 9**,** 65-79.
GOLDSTEIN, J. H., PRESNALL, C. K., LOPEZ-HOFFMAN, L., NABHAN, G. P., KNIGHT, R. L., RUYLE, G. B. & TOOMBS, T. P. (2011) Beef and Beyond: Paying for Ecosystem Services on Western US Rangelands. *Rangelands,* 33**,** 4-12.
GÓMEZ-BAGGETHUN, E., DE GROOT, R., LOMAS, P. L. & MONTES, C. (2010) The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics,* 69**,** 1209-1218.
GÓMEZ-BAGGETHUN, E. & RUIZ-PEREZ, M. (2011) Economic valuation and the commodification of ecosystem services. *Progress in Physical Geography,* 35**,** 613-628.
GOODMAN, J. C. & STROUP, R. L. (1991) Progressive Environmentalism: A Pro-Human, Pro-Science, Pro-Free Enterprise Agenda for Change. *Environmental Task Force Report.* Mackinac Center for Public Policy.
GOVERNMENT OF KENYA (2003) Economic Recovery Strategy for Wealth and Employment Creation 2003-2007. Nairobi, Ministry of Planning and National Development.
GOVERNMENT OF KENYA (2007) Draft Wildlife Bill (2007). IN MINISTRY OF TOURISM & WILDLIFE (Ed.). Nairobi, Government Publishers.
GOVERNMENT OF KENYA (2008a) Kenya Vision2030: First Medium Term Plan 2008-2012. Nairobi, Government Printers.
GOVERNMENT OF KENYA (2008b) Nairobi Metro 2030: A World Class African Metropolis. Nairobi, Ministry of Nairobi Metropolitan Development (MNMD), Government of Kenya.
GOVERNMENT OF KENYA (2010) National Climate Change Response Strategy. Nairobi, Ministry of Environment and Mineral Resources, Government of Kenya (GoK).
GREEN, W. (2003) *Econometric Analsysis (5th Edition),* New Jersey, Prentice Hall.
GREINER, R., GORDON, I. & COCKLIN, C. (2009) Ecosystem services from tropical savannas: economic opportunities through payments for environmental services. *The Rangeland Journal,* 31**,** 51-59.
GREINER, R. & STANLEY, O. (2013) More than money for conservation: Exploring social co-benefits from PES schemes. *Land Use Policy,* 31**,** 4-10.
GRIEG-GRAN, M., PORRAS, I. & WUNDER, S. (2005) How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. *World Development,* 33**,** 1511-1527.
HACKEL, J. D. (1999) Community conservation and the future of Africa's wildlife. *Conservation Biology,* 13**,** 726-734.

HANSEN, A. J. & DEFRIES, R. (2007) Land Use Change around Nature Reserves: Implications for Sustaining Biodiversity. *Ecological Applications,* 17**,** 972-973.
HERTEL, T. W. & ROSCH, S. D. (2010) Climate Change, Agriculture, and Poverty. *Applied Economic Perspectives and Policy,* 32**,** 355-385.
HOBBS, N. T., GALVIN, K. A., STOKES, C. J., LACKETT, J. M., ASH, A. J., BOONE, R. B., REID, R. S. & THORNTON, P. K. (2008a) Fragmentation of rangelands: Implications for humans, animals, and landscapes. *Global Environmental Change,* 18**,** 776-785.
HOBBS, N. T., REID, R. S., GALVIN, K. A. & ELLIS, J. E. (2008b) Fragmentation of Arid and Semi-Arid Ecosystems: Implications for People and Animals. IN GALVIN, K. A., REID, R. S., BEHNKE, R. & HOBBS, N. T. (Eds.) *Fragmentation in Semi-Arid Landscapes: Consequences for Human and Natural Systems.* Springer.
HOMEWOOD, K. (2004) Policy, environment and development in African rangelands. *Environmental Science & Policy,* 7**,** 125-143.
HOMEWOOD, K. (2008) *Ecology of African Pastoralist Societies,* James Currey Ltd, Oxford.
HOMEWOOD, K. (2009) Policy and Practice in Kenya Rangelands: Impacts on Livelihoods and Wildlife. IN HOMEWOOD, K., KRISTJANSON, P. & TRENCH, P. (Eds.) *Staying Maasai? Livelihoods, Conservation and Development in East African Rangelands.* Springer.
HOMEWOOD, K., KRISTJANSON, P. & TRENCH, P. (2009a) Changing Land Use, Livelihoods and Wildlife Conservation in Maasailand. IN HOMEWOOD, K., KRISTJANSON, P. & TRENCH, P. (Eds.) *Staying Maasai? Livelihoods, Conservation and Development in East African Rangelands.* Springer.
HOMEWOOD, K., KRISTJANSON, P. & TRENCH, P. (2009b) Staying Maasai? Pastoral Livelihoods, Diversification and the Role of Wildlife in Development. IN HOMEWOOD, K., KRISTJANSON, P. & TRENCH, P. (Eds.) *Staying Maasai? Livelihoods, Conservation and Development in East African Rangelands.* Springer.
HOMEWOOD, K., KRISTJANSON, P. & TRENCH, P., C. (Eds.) (2009c) *Staying Maasai? Livelihoods, Conservation and Development in East Africa's Rangelands*, Springer.
HOMEWOOD, K., LAMBIN, E. F., COAST, E., KARIUKI, A., KIKULA, I., KIVELIA, J., SAID, M., SERNEELS, S. & THOMPSON, M. (2001) Long-term changes in Serengeti-Mara wildebeest and land cover: Pastoralism, population, or policies? *Proceedings of the National Academy of Sciences*, 98**,** 12544-12549.
HOMEWOOD, K., TRENCH, P. & BROCKINGTON, D. (2012) Pastoralist livelihoods and wildlife revenues in East Africa: a case for coexistence? *Pastoralism: Research, Policy and Practice,* 2**,** 19.
HONEY, M. (2009) Community Conservation and Early Ecotourism: Experiments in Kenya. *Environment,* 51**,** 47-56.
HOPCRAFT, D. (2000) Wildlife Land Use and the Great Experiment. IN PRINS, H. H. T., GROOTENHUIS, J. G. & DOLAN, T. T. (Eds.) *Wildlife Conservation by Sustainable Use.* Kluwer Academic Publishers.
HORAN, R. D., SHOGREN, F. F. & GRAMIG, B. M. (2008) Wildlife conservation payments to address habitat fragmentation and disease risks. *Environment and Development Economics,* 13**,** 415-439.
HUBERMAN, D. (2009) World Heritage Sites and the Integration of Biodiversity into an International PES Framework. *Journal of Sustainable Forestry,* 28**,** 456 - 480.
HULME, D. & MURPHREE, M. (1999) Communities, wildlife and the 'new conservation' in Africa. *Journal of International Development,* 11**,** 277-285.
HURST, M., JENSEN, N., PEDERSEN, S. H., SHARMA, A. & ZAMBRISKI, J. A. (2012) Changing climate adaptation strategies of Boran pastoralists in southern Ethiopia.

Working paper no. 15. Cali, Colombia, CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS).

IFTIKHAR, U., KALLESOE, M., DURAIAPPAH, A. K., SRISKANTHAN, G., POATS, S. & SWALLOW, B. M. (2007) Exploring the inter-linkages among and between Compensation and Rewards for Ecosystem Services (CRES) and human well-being. *ICRAF Working Paper Series no. 36.* Nairobi, ICRAF.

IKIARA, M. (2001) Policy Framework of Kenya's Tourism Sector since Independence and Emerging Policy Concerns. *KIPPRA Working Paper No. 2.* Nairobi, Kenya, Kenya Institute for Public Policy Research and Analysis (KIPPRA).

IKIARA, M. & OKECH, C. (2002) Impact of Tourism on Environment in Kenya: Status and Policy. *Discussion Paper No. 19.* Nairobi, Kenya, Kenya Institute for Public Policy Research and Analysis (KIPPRA).

IMBAHALE, S. S., GITHAIGA, J. M., CHIRA, R. M. & SAID, M. Y. (2008) Resource utilization by large migratory herbivores of the Athi-Kapiti ecosystem. *African Journal of Ecology,* 46**,** 43-51.

INTERNATIONAL MONETARY FUND (2005) Kenya: Poverty Reduction Strategy Paper. *IMF Country Report No. 05/11 (January 2005).* Washington, D.C., The International Monetary Fund (IMF).

JACK, B. K., KOUSKY, C. & SIMS, K. R. E. (2008) Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms. *Proceedings of the National Academy of Sciences of the United States of America,* 105**,** 9465-9470.

JANSSEN, M. A. & OSTROM, E. (2006) Resilience, vulnerability, and adaptation: A cross-cutting theme of international human dimensions programme on global environmental change. *Global Environmental Change-Human and Policy Dimensions,* 16**,** 237-239.

JENKINS, M., SCHERR, S. J. & INBAR, M. (2004) Markets for biodiversity services - Potential roles and challenges. *Environment,* 46**,** 32-42.

JINDAL, R., SWALLOW, B. M. & KERR, J. (2008) Forestry-based carbon sequestration projects in Africa: Potential benefits and challenges. *Natural Resources Forum,* 32**,** 116-130.

KABIRI, N. (2010) The Political Economy of Wildlife Conservation and Decline in Kenya. *The Journal of Environment & Development,* 19**,** 424-445.

KABUBO-MARIARA, J. (2005) Herders response to acute land pressure under changing property rights: some insights from Kajiado District, Kenya. *Environment and Development Economics,* 10**,** 67-85.

KABUBO-MARIARA, J. (2009) Global warming and livestock husbandry in Kenya: Impacts and adaptations. *Ecological Economics,* 68**,** 1915-1924.

KAESLIN, E., REDMOND, I. & DUDLEY, N. (2012) Wildlife in a changing climate. Rome, FAO (The Food and Agriculture Organisation of the United Nations).

KAMERI-MBOTE, P. (2002) *Property Rights and Biodiversity Management in Kenya: The Case of Land Tenure and Wildlife,* Nairobi, ACTS Press.

KAMERI-MBOTE, P. (2005) Land Tenure, Land Use and Sustainability in Kenya: Towards Innovative Use of Property Rights In Wildlife Management. International Environmental Law Research Center (IELRC).

KAREITHI, S. (2003) Coping with Declining Tourism, Examples from Communities in Kenya. *Pro-Poor Tourism (PPT) Working Paper No. 13.* Pro-Poor Tourism Partnership (ODI/ICRT/IIED).

KAREIVA, P., CHANG, A. & MARVIER, M. (2008) Development and Conservation Goals in World Bank Projects. *Science,* 321**,** 1638-1639.

KELLERT, S. R., MEHTA, J. N., EBBIN, S. A. & LICHTENFELD, L. L. (2000) Community natural resource management: Promise, rhetoric, and reality. *Society & Natural Resources,* 13**,** 705-715.

KENYA NATIONAL BUREAU OF STATISTICS (2009) STATISTICAL ABSTRACT 2009. Nairobi, Kenya National Bureau of Statistics (KNBS).

KENYA NATIONAL BUREAU OF STATISTICS (2010) The 2009 Kenya Population and Housing Census. Counting Our People for the Implementation of Vision 2030. Volume 1A. Population Distribution by Administrative Units. Nairobi, Kenya National Bureau of Statistics (KNBS).

KENYA NATIONAL BUREAU OF STATISTICS (2011) STATISTICAL ABSTRACT 2011. Nairobi, Kenya National Bureau of Statistics (KNBS).

KEPHER-GONA, J. (2006) The Great Mara Debate: surviving challenges of conservation and development. *Proceedings of a Workshop held on September 21, 2006, Holiday Inn.* Nairobi, Ecotourism Kenya.

KERR, J. (2002) Watershed development, environmental services, and poverty alleviation in India. *World Development,* 30**,** 1387-1400.

KHISA, K. (2001) Testing of Techniques for Resolving Conflicts in Natural Resource Management: The Case of Nairobi National Park in Kenya. *Report to the UNESCO Man & Biosphere Programme.* Paris, UNESCO.

KIMANI, K. & PICKARD, J. (1998) Recent trends and implications of group ranch sub-division and fragmentation in Kajiado District. *Geographical Journal,* 164**,** 202.

KING, K. & PALMER, R. (2007) Skills, Development and Poverty Reduction: A State of the Art Review. Turin, Italy, European Training Forum.

KINZIG, A. P., PERRINGS, C., CHAPIN, F. S., POLASKY, S., SMITH, V. K., TILMAN, D. & TURNER, B. L. (2011) Paying for Ecosystem Services - Promise and Peril. *Science,* 334**,** 603-604.

KISHORE, S. (2007) Mainstreaming Environment in the Implementation of PRSPs in Sub-Saharan Africa. *Environment Department Working Paper No. 112.* Washington, D.C., World Bank.

KISS, A. (1990) Living with Wildlife: Wildlife Resource Management with Local Participation in Africa. *World Bank Technical Paper No. 130.* Washington D.C., World Bank.

KISS, A. (2004a) Is community-based ecotourism a good use of biodiversity conservation funds? *Trends in Ecology & Evolution,* 19**,** 232-237.

KISS, A. (2004b) Making Biodiversity Conservation a Land-Use Priority. IN MCSHANE, T. O. & WELLS, M. (Eds.) *Getting Biodiversity Projects to Work.* New York, Columbia University Press.

KOSOY, N. & CORBERA, E. (2009) Payments for ecosystem services as commodity fetishism. *Ecological Economics,* 69**,** 1228-1236.

KOSOY, N., CORBERA, E. & BROWN, K. (2008) Participation in payments for ecosystem services: Case studies from the Lacandon rainforest, Mexico. *Geoforum,* 39**,** 2073-2083.

KOSOY, N., MARTINEZ-TUNA, M., MURADIAN, R. & MARTINEZ-ALIER, J. (2007) Payments for environmental services in watersheds: Insights from a comparative study of three cases in Central America. *Ecological Economics,* 61**,** 446-455.

KREMU (1983) Land use map of Kenya. Nairobi, KREMU (Kenya Rangeland Ecological Monitoring Unit).

KRISTJANSON, P., RADENY, M., BALTENWECK, I., OGUTU, J. O. & NOTENBAERT, A. (2005) Livelihood mapping and poverty correlates at a meso-level in Kenya. *Food Policy,* 30**,** 568-583.

KRISTJANSON, P., RADENY, M., NKEDIANYE, D., KRUSKA, R. L., REID, R. S., GICHOHI, W. H., ATIENO, F. & SANFORD, R. (2002) Valuing alternative land-use options in the Kitengela wildlife dispersal area of Kenya. *ILRI Impact Assessment Series 10.* Nairobi, International Livestock Research Center (ILRI)/ACC (Africa Conservation Center).

KWS (2004) Nairobi National Park Ecosystem Management Plan: 2003-2008. Nairobi, Kenya Wildlife Service (KWS).

KWS (2012) CORPORATE POLICY DOCUMENT: Policy on Establishment of Conservancies, Training and Management of County Reserve, Private and Community Rangers. *Corporate Policy Document (1st edition).* Nairobi, Kenya Wildlife Service (KWS).

LAMBIN, E. F. & MEYFROIDT, P. (2011) Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Sciences,* 108**,** 3465-3472.

LAMPREY, R. H. & REID, R. S. (2004) Expansion of human settlement in Kenya's Maasai Mara: what future for pastoralism and wildlife? *Journal of Biogeography,* 31**,** 997-1032.

LANDELL-MILLS, N. (2002) Developing Markets for Forest Environmental Services: An Opportunity for Promoting Equity While Securing Efficiency? *Philosophical Transactions: Mathematical, Physical and Engineering Sciences,* 360**,** 1817-1825.

LANE, R. C. (Ed.) (1998) *Custodians of the Commons: Pastoral Land Tenure in East and West Africa,* London, Earthscan Publications.

LANGHOLZ, J. A. & KRUG, W. (2004) NEW FORMS OF BIODIVERSITY GOVERNANCE: NON-STATE ACTORS AND THE PRIVATE PROTECTED AREA ACTION PLAN. *Journal of International Wildlife Law & Policy,* 7**,** 9-29.

LARIGAUDERIE, A. & MOONEY, H. A. (2010) The Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services: moving a step closer to an IPCC-like mechanism for biodiversity. *Current Opinion in Environmental Sustainability,* 2**,** 9-14.

LEE, E. & MAHANTY, S. (2009) Payments for Environmental Services and Poverty Reduction: Risks and Opportunities. Bankok, Thailand, The Center for People and Forests (RECOFT).

LEIMONA, B. & JOSHI, L. (2009) Can rewards for environmental services benefit the poor? Lessons from Asia. *International Journal of the Commons,* 3**,** 82-107.

LENAOLA, I., JENNER, H. H. & WICHERT, T. (1996) Land tenure in pastoral lands. IN JUMA, C. & OJWANG, J. B. (Eds.) *In Land We Trust: Environment, Private Property and Constitutional Change.* Nairobi, Kenya and London, UK, Initiatives Publishers and Zed Books.

LI, J., FELDMAN, M. W., LI, S. & DAILY, G. C. (2011) Rural household income and inequality under the Sloping Land Conversion Program in western China. *Proceedings of the National Academy of Sciences,* 108**,** 7721-7726.

LINDSEY, P. A., ROULET, P. A. & ROMAÑACH, S. S. (2007) Economic and conservation significance of the trophy hunting industry in sub-Saharan Africa. *Biological Conservation,* 134**,** 455-469.

LIPPER, L., SAKUYAMA, T., STRINGER, R. & ZILBERMAN, D. (2009) *Payment for Environmental Services in Agricultural Landscapes: Economic Policies and Poverty Reduction in Developing Countries,* Springer.

LITTLE, P. D., MCPEAK, J., BARRETT, C. B. & KRISTJANSON, P. (2008) Challenging Orthodoxies: Understanding Poverty in Pastoral Areas of East Africa. *Development and Change,* 39**,** 587-611.

LOH, J. & WACKERNAGEL, M. (2004) Living Plant Report. Gland, Switzerland, WWF (World Wide Fund for Nature)/UNEP World Conservation Monitoring Center/Global Footprint Network.
LUCK, G. W., CHAN, K. M. A., ESER, U., GÓMEZ-BAGGETHUN, E., MATZDORF, B., NORTON, B. & POTSCHIN, M. B. (2012) Ethical Considerations in On-Ground Applications of the Ecosystem Services Concept. *Bioscience,* 62**,** 1020-1029.
LYBBERT, T. J., BARRETT, C. B., DESTA, S. & LAYNE COPPOCK, D. (2004) Stochastic wealth dynamics and risk management among a poor population. *The Economic Journal,* 114**,** 750-777.
MANYARA, G. & JONES, E. (2007) Community-based Tourism Enterprises Development in Kenya: An Exploration of Their Potential as Avenues of Poverty Reduction. *Journal of Sustainable Tourism,* 15**,** 628-644.
MARTIN, A., BLOWERS, A. & BOERSEMA, J. (2008) Paying for environmental services: can we afford to lose a cultural basis for conservation? *Journal of Integrative Environmental Sciences,* 5**,** 1-5.
MCAFEE, K. (2012) The Contradictory Logic of Global Ecosystem Services Markets. *Development and Change,* 43**,** 105-131.
MCAFEE, K. & SHAPIRO, E. N. (2010) Payments for Ecosystem Services in Mexico: Nature, Neoliberalism, Social Movements, and the State. *Annals of the Association of American Geographers,* 100**,** 579-599.
MCCARTHY, J., CANZIANI, O. F., LEARY, N. A., DOKKEN, D. J. & WHITE, C. (Eds.) (2001) *Climate change 2001: impacts, adaptation and vulnerability,* Cambridge, UK, Cambridge University Press and IPCC (Intergovernmental Panel on Climate Change).
MCCAULEY, D. J. (2006) Selling out on nature. *Nature,* 443**,** 27-28.
MCDERMOTT, M., MAHANTY, S. & SCHRECKENBERG, K. (2012) Examining equity: A multidimensional framework for assessing equity in payments for ecosystem services. *Environmental Science & Policy***,** DOI: http://dx.doi.org/10.1016/j.envsci.2012.10.006.
MCPEAK, J., LITTLE, P. D. & DOSS, C. (2012) *Risk and Social Change in an African Rural Economy,* London, UK and New York, USA, Routledge (Taylor & Francis Group).
MEIJERINK, G. (2008) The Role of Measurement Problems And Monitoring In PES Schemes IN DELLINK, R. B. & RUIJS, A. (Eds.) *Economics of Poverty, Environment and Natural-Resource Use.* Springer Netherlands.
MILDER, J. C., SCHERR, S. J. & BRACER, C. (2010) Trends and Future Potential of Payment for Ecosystem Services to Alleviate Rural Poverty in Developing Countries. *Ecology and Society,* 15 (2).
MILLENNIUM ECOSYSTEM ASSESSMENT (2005a) Ecosystems and Human Well-being: Biodiversity Synthesis. Washington DC, World Resources Institute (WRI).
MILLENNIUM ECOSYSTEM ASSESSMENT (2005b) Ecosystems and Human Well-being: Synthesis. Washington D.C., World Resources Institute (WRI).
MILNE, S. & NIESTEN, E. (2009) Direct payments for biodiversity conservation in developing countries: practical insights for design and implementation. *Oryx,* 43**,** 530-541.
MINISTRY OF LANDS (2007) NATIONAL LAND POLICY. Nairobi, Kenya, Ministry of Lands (MoL), National Land Policy Secretariat.
MIRANDA, M. E. G., PORRAS, I. & MORENO, M. L. (2003) The social impacts of payments for environmental services in Costa Rica. A quantitative field survey and analysis of the Virilla watershed. London, International Institute for Environment and Development (IIED).
MORTIMORE, M., ANDERSON, S., COTULA, L., DAVIES, J., FACCER, K., HESSE, C., MORTON, J., NYANGENA, W., SKINNER, J. & WOLFANGEL, C. (2009) Dryland

Opportunities: A new paradgim for people, ecosystems and development. IUCN/IIED/UNDP-DDC.
MUDE, A., OUMA, R., VAN DE STEEG, J., KARIUKI, J., OPIYO, D. & TIPILDA, A. (2007) Kenya adaptation to climate change in the arid lands: Anticipating, adapting to and coping with climate risks in Kenya - Operational recommendations for KACCAL. *ILRI Research Report No. 18.* Nairobi, Kenya, International Livestock Research Institute.
MUIR-LERESCHE, K. & NELSON, H. R. (2000) Private Property Rights to Wildlife: The Southern African Experiment. Washington, D.C., Competetive Enterprise Institute (CEI).
MUIR-LERESCHE, K. & NELSON, R. H. (2001) Managing Wildlife in Southern Africa. *PERC Reports.* Bozeman, MT, PERC (Center for Free Market Environmentalism).
MULLAN, K. & KONTOLEON, A. (2009) Participation in Payments for Ecosystem Service programmes in developing countries: The Chinese Sloping Land Conversion Programme. *Environmental Economy and Policy Research Discussion Paper Series No. 42.2009.* Cambridge, U.K., University of Cambridge, Department of Land Economics.
MULLER, J. & ALBERS, H. J. (2004) Enforcement, payments, and development projects near protected areas: how the market setting determines what works where. *Resource and Energy Economics,* 26**,** 185-204.
MUNOZ-PINA, C., GUEVARA, A., TORRES, J. M. & BRANA, J. (2008) Paying for the hydrological services of Mexico's forests: Analysis, negotiations and results. *Ecological Economics,* 65**,** 725-736.
MURADIAN, R., CORBERA, E., PASCUAL, U., KOSOY, N. & MAY, P. H. (2010) Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services. *Ecological Economics,* 69**,** 1202-1208.
MWANGI, E. (2007a) The Puzzle of Group Ranch Subdivision in Kenya's Maasailand. *Development and Change,* 38**,** 889-910.
MWANGI, E. (2007b) Subdividing the Commons: Distributional Conflict in the Transition from Collective to Individual Property Rights in Kenya's Maasailand. *World Development,* 35**,** 815-834.
MWANGI, E. & OSTROM, E. (2009a) A Century of Institutions and Ecology in East Africa's Rangelands: Linking Institutional Robustness with the Ecological Resilience of Kenya's Maasailand. IN BECKMANN, V. & PADMANABHAN, M. (Eds.) *Institutions and Sustainability: Political Economy of Agriculture and the Environment - Essays in Honour of Konrad Hagedorn.* Springer Verlag.
MWANGI, E. & OSTROM, E. (2009b) Top-Down Solutions: Looking Up from East Africa's Rangelands. *Environment,* 51**,** 34-45.
MWANGI, S. & WARINDA, E. (1999) Socio-Economic Dimensions of Sustainable Wildlife Conservation in the Kitengela Dispersal Area. Nairobi, ACC (Africa Conservation Center).
NAIDOO, R., WEAVER, L. C., DE LONGCAMP, M. & DU PLESSIS, P. (2011) Namibia's community-based natural resource management programme: an unrecognized payments for ecosystem services scheme. *Environmental Conservation,* 38**,** 445-453.
NASSEF, M., ANDERSON, S. & HESSE, C. (2009) Pastoralism and climate change: Enabling adaptive capacity. HPG (Humanitarian Policy Group).
NAUGHTON-TREVES, L., HOLLAND, M. B. & BRANDON, K. (2005) The Role of Protected Areas in Conserving Biodiversity and Sustaining Local Livelihoods. *Annual Review of Environment and Resources,* 30**,** 219-252.
NAUGHTON-TREVES, L. & SANDERSON, S. E. (1995) Property, politics and wildlife conservation. *World Development,* 23**,** 1265-1275.

NAVRUD, S. & MUNGATANA, E. D. (1994) Environmental valuation in developing countries: The recreational value of wildlife viewing. *Ecological Economics,* 11**,** 135-151.

NCC & TCC (2009) Maasai Mara National Reserve Management Plan 2009-2019 (2nd Draft, March 2009). NCC (Narok County Council) & TCC (Trans-Mara County Council) with support of the Africa Wildlife Foundation (AWF), the Mara Conservancy and the Conservation Development Center (CDC).

NEELY, C. & BUNNING, S. (2008) Review of Evidence on Dryland Pastoral Systems and Climate Change: Implications and opportunities for mitigation and adaptation. *FAO-NRL Working Paper.* Rome, Italy, FAO (Food and Agricultural Organisation of the United Nations).

NELSON, D. R., ADGER, W. N. & BROWN, K. (2007) Adaptation to Environmental Change: Contributions of a Resilience Framework. *Annual Review of Environment and Resources,* 32**,** 395-419.

NELSON, F. (2009) Developing Payments for Ecosystem Services Approaches to Carnivore Conservation. *Human Dimensions of Wildlife,* 14**,** 381-392.

NELSON, F., FOLEY, C., FLOEY, L. S., LEPOSO, A., LOURE, E., PETERSON, D., PETERSON, M., PETERSON, T., SACHEDINA, H. & WILLIAMS, A. (2010) Payments for Ecosystem Services as a Framework for Community-Based Conservation in Northern Tanzania. *Conservation Biology,* 24**,** 78-85.

NEWMARK, W. D. (2008) Isolation of African protected areas. *Frontiers in Ecology and the Environment,* 6**,** 321-328.

NEWMARK, W. D. & HOUGH, J. L. (2000) Conserving Wildlife in Africa: Integrated Conservation and Development Projects and Beyond. *Bioscience,* 50.

NIAMIR-FULLER, M. (1999) Managing Mobility in African Rangelands. IN MCCARTHY, N., SWALLOW, B. M., KIRK, M. & HAZELL, P. (Eds.) *Property Rights, Risks and Livestock Development in Africa.* IFPRI (International Food Policy Institute) & ILRI (International Livestock Research Institute).

NKEDIANYE, D. (2004) Testing the Attitudinal Impact of A Conservation Tool Outside A Protected Area: The Case of the Kitengela Wildlife Conservation Lease Programme for Nairobi National Park. *Sociology.* Nairobi, Masters Thesis, University of Nairobi.

NKEDIANYE, D., DE LEEUW, J., OGUTU, J. O., SAID, M. Y., SAIDIMU, T., KIFUGO, S., KAELO, D. & REID, R. S. (2011) Mobility and livestock mortality in communally used pastoral areas: the impact of the 2005-2006 drought on livestock mortality in Maasailand. *Pastoralism: Research, Policy and Practice,* 1**,** 17.

NKEDIANYE, D., RADENY, M., KRISTJANSON, P. & HERRERO, M. (2009) Assessing Returns to Land and Changing Livelihood Strategies in Kitengela. IN HOMEWOOD, K., KRISTJANSON, P. & TRENCH, P. C. (Eds.) *Staying Maasai? Livelihoods, Conservation and Development in East African Rangelands.* Springer.

NORI, M. & DAVIES, J. (2007) Change of Wind or Wind of Change? Climate change, adaptation and pastoralism. Nairobi, WISP (World Initiative for Sustainable Pastoralism).

NORTON-GRIFFITHS, M. (1996) Property rights and the marginal wildebeest: An economic analysis of wildlife conservation options in Kenya. *Biodiversity and Conservation,* 5**,** 1557-1577.

NORTON-GRIFFITHS, M. (1998) The economics of wildlife conservation policy in Kenya. IN MILNER-GULLAND, E. J. & MACE, R. (Eds.) *Conservation of Biological Resources.* Oxford, UK, Blackwell Science.

NORTON-GRIFFITHS, M. (2007a) How Many Wildebeest do You Need? *World Economics,* 8**,** 41-64.

NORTON-GRIFFITHS, M. (2007b) Whose wildlife is it anyway? *New Scientist,* 193**,** 24-24.
NORTON-GRIFFITHS, M., SAID, M. Y., SERNEELS, S., KAELO, D. S., COUGHENOUR, M., LAMPREY, R. H., THOMPSON, D. M. & REID, R. S. (2008) Land Use Economics in the Mara Area of the Serengeti Ecosystem. IN SINCLAIR, A. R. E., PARKER, C., MDUMA, S. A. R. & FRYXELL, J. M. (Eds.) *Serengeti III: Human Impacts on Ecosystem Dynamics.* Chicago, University of Chicago Press.
NORTON-GRIFFITHS, M. & SAID, Y. M. (2010) The Future for Wildlife on Kenya's Rangelands: An Economic Perspective. IN DU TOIT, J. T., KOCK, R. & DEUTSCH, J. C. (Eds.) *Wild Rangelands: Conserving Wildlife While Maintaining Livestock in Semi-Arid Ecosystems.* Blackwell Publishing.
NORTON-GRIFFITHS, M. & SOUTHEY, C. (1995) The opportunity costs of biodiversity conservation in Kenya. *Ecological Economics,* 12**,** 125-139.
O'MEARA, R. (2011) PES Scheme Intermediary Interview: the Olare Orok Conservancy (OOC).
ODI (OVERSEAS DEVELOPMENT INSTITUTE) (2006) Can Tourism Help Reduce Poverty in Africa? *ODI Briefing Paper March 2006.* London, UK, ODI (Overseas Development Institute).
OECD-DAC (2001) Poverty Reduction Guidelines. Paris, OECD-DAC (Organisation for Economic Co-operation and Development, Development Assistance Committee).
OECD (2003) Harnessing Markets for Biodiversity:Towards Conservation and Sustainable Use. Paris: OECD (Organisation for Economic Co-operation and Development).
OECD (2004) Handbook of Market Creation for Biodiversity: Issues in Implementation. Paris: OECD (Organisation for Economic Co-operation and Development).
OECD (2010) Paying for Biodiversity: Enhancing the Cost-Effectiveness of Payment for Ecosystem Services. Paris, Organisation of Economic Co-operation and Development (OECD).
OGUTU, J. O., OWEN-SMITH, N., PIEPHO, H. P. & SAID, M. Y. (2011) Continuing wildlife population declines and range contraction in the Mara region of Kenya during 1977–2009. *Journal of Zoology,* 285**,** 99-109.
OGUTU, J. O., PIEPHO, H. P., DUBLIN, H. T., BHOLA, N. & REID, R. S. (2007) El Niño-Southern Oscillation, rainfall, temperature and Normalized Difference Vegetation Index fluctuations in the Mara-Serengeti ecosystem. *African Journal of Ecology,* 46**,** 132-143.
OGUTU, Z. A. (2002) The impact of ecotourism on livelihood and natural resource management in Eselenkei, Amboseli Ecosystem, Kenya. *Land Degradation & Development,* 13**,** 251-256.
OKELLO, M. M. (2005) Land Use Changes and Human-Wildlife Conficts in the Amboseli Area, Kenya. *Human Dimensions of Wildlife,* 10**,** 19-28.
OKELLO, M. M., OLE SENO, S. K. & NTHIGA, R. W. (2009) Reconciling people's livelihoods and environmental conservation in the rural landscapes in Kenya: Opportunities and challenges in the Amboseli landscapes. *Natural Resources Forum,* 33**,** 123-133.
OKWI, P. O., NDENG'E, G., KRISTJANSON, P., ARUNGA, M., NOTENBAERT, A., OMOLO, A., HENNINGER, N., BENSON, T., KARIUKI, P. & OWUOR, J. (2007) Spatial determinants of poverty in rural Kenya. *Proceedings of the National Academy of Sciences of the United States of America,* 104**,** 16769-16774.
OMITI, J. & IRUNGU, P. (2002) Institutional and Policy Issues Relevant to Pastoral Development in Kenya. Nairobi, IPAR (Institute of Policy Analysis and Research).
OMONDI, P. (1994) Wildlife-Human Conflict in Kenya: Integrating Wildlife Conservation with Human Needs in the Masai Mara Region. *Geography Department.* Montreal, Canada, PhD Thesis, McGill University.

ORINDI, A. V., NYONG, A. & HERRERO, M. (2007) Pastoral Livelihood Adaptation to Drought and Institutional Interventions in Kenya. *Human Development Report Office Ocassional Paper 2007/54.* UNDP (United Nations Development Programme).

OSANO, P. (2011) Life at the Crossroads: How Climate Change Threatens the Existence of the Maasai. Waterloo, ON, Canada, Center for International Governance Innovation (CIGI).

OSANO, P. (2012) Improving Policy Response and Finding Sustainable Solutions to Drought in Kenya. Waterloo, Ontario, Canada, Center for International Governance Innovation (CIGI).

OTTICHILO, K. W., DE LEEUW, J., SKIDMORE, A. K., PRINS, H. H. T. & SAID, Y. M. (2000) Population trends of large non-migratory wild herbivores and livestock in the Masai Mara ecosystem, Kenya, between 1977 and 1997. *African Journal of Ecology,* 38**,** 202-216.

PAGIOLA, S. (2003) Can Programs of Payment for Environmental Services Help Preserve Wildlife? *CITES (The Convention on International Trade on Endangered Species of Wild Fauna and Flora) Workshop on Economic Incentives and Trade Policies, 1-3 December, 2003, Geneva, Switzerland.* Geneva, Switzerland, CITES (The Convention on International Trade on Endangered Species of Wild Fauna and Flora.

PAGIOLA, S. (2007) Guidelines for "Pro-Poor" Payments for Environmental Services. Washington, D.C., World Bank (Environment Department).

PAGIOLA, S. (2008) Payments for environmental services in Costa Rica. *Ecological Economics,* 65**,** 712-724.

PAGIOLA, S., ARCENAS, A. & PLATAIS, G. (2005) Can Payments for Environmental Services Help Reduce Poverty? An Exploration of the Issues and the Evidence to Date from Latin America. *World Development,* 33**,** 237-253.

PAGIOLA, S., RAMÍREZ, E., GOBBI, J., DE HAAN, C., IBRAHIM, M., MURGUEITIO, E. & RUÍZ, J. P. (2007) Paying for the environmental services of silvopastoral practices in Nicaragua. *Ecological Economics,* 64**,** 374-385.

PAGIOLA, S., RIOS, A. & ARCENAS, A. (2010) Poor Household Participation in Payments for Environmental Services: Lessons from the Silvopastoral Project in Quindío, Colombia. *Environmental and Resource Economics,* 47**,** 371-394.

PAGIOLA, S., RIOS, A. R. & ARCENAS, A. (2008) Can the poor participate in payments for environmental services? Lessons from the Silvopastoral Project in Nicaragua. *Environment and Development Economics,* 13**,** 299-325.

PASCUAL, U., MURADIAN, R., RODRÍGUEZ, L. C. & DURAIAPPAH, A. K. (2010) Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. *Ecological Economics,* 69**,** 1237-1244.

PATTANAYAK, S. K., WUNDER, S. & FERRARO, P. J. (2010) Show Me the Money: Do Payments Supply Environmental Services in Developing Countries? *Review of Environmental Economics and Policy,* 4**,** 254-274.

PEARCE, D. (2005) The Critical Role of Environmental Improvement in Poverty Reduction. *Report Prepared for the Poverty Environment Partnership.* United Nations Development Programme.

PEREZ, C., RONCOLI, C., NEELY, C. & STEINER, J. L. (2007) Can carbon sequestration markets benefit low-income producers in semi-arid Africa? Potentials and challenges. *Agricultural Systems,* 94**,** 2-12.

PERRINGS, C., DURAIAPPAH, A. K., LARIGAUDERIE, A. & MOONEY, H. (2011) The Biodiversity and Ecosystem Services Science-Policy Interface. *Science,* 331**,** 1139-1140.

PERRY, B. & SONES, K. (2007) Poverty Reduction Through Animal Health. *Science,* 315**,** 333-334.

PERSSON, M. U. & ALPIZAR, F. (2013) Conditional Cash Transfers and Payments for Environmental Services - A Conceptual Framework for Explaining and Judging Differences in Outcomes. *World Development,* 43**,** 124-137.

PORRAS, I., GRIEG-GRAN, M. & NEVES, N. (2008) All that glitters: A review of payments for watershed in developing countries. London, UK, International Institute for Environment and Development (IIED).

POWER, A. G. (2010) Ecosystem services and agriculture: tradeoffs and synergies. *Philosophical Transactions of the Royal Society B: Biological Sciences,* 365**,** 2959-2971.

PREGIBON, D. (1981) Logistic Regression Diagnostics. *The Annals of Statistics,* 9**,** 705-724.

PURVIS, A. & HECTOR, A. (2000) Getting the measure of biodiversity. *Nature,* 405**,** 212-219.

RADENY, M., NKEDIANYE, D., KRISTJANSON, P. & HERRERO, M. (2007) Livelihood Choices and Returns Among Pastoralists: Evidence from Southern Kenya. *Nomadic Peoples,* 11**,** 31-55.

RAFFMAN DHANJI ELMS VIRDEE ADVOCATES (2011) Olare Orok Conservancy Ltd Lease. *Ref M24301E/MO.* Nairobi.

RANDALL, S. (2008) African Pastoralist Demography. IN HOMEWOOD, K. (Ed.) *Ecology of African Pastoralist Societies.* Oxford, James Currey Ltd.

RAVNBORG, H. M., DAMSGAARD, M. G. & RABEN, K. (2007) Payments for Ecosystem Services - Issues and Pro-Poor Opportunities for Development Assistance. Copenhagen, Danish Institute for International Studies.

REARDON, T. & VOSTI, S. A. (1995) Links between rural poverty and the environment in developing countries: Asset categories and investment poverty. *World Development,* 23**,** 1495-1506.

REDFORD, K. H. & FEARN, E. (2007) Protected Areas and Human Livelihoods. *WCS Working Papers.* Bronx, New York, Wildlife Conservation Society (WCS).

REID, R. S., GICHOHI, W. H., SAID, M. Y., NKEDIANYE, D., OGUTU, J. O., KSHATRIYA, M., KRISTJANSON, P., KIFUGO, S., AGATSIVA, J. L., ADANJE, S. A. & BAGINE, R. (2008) Fragmentation of A Peri-Urban Savanna, Athi-Kaputie Plains, Kenya. IN GALVIN, K. A., REID, R. S., BEHNKE, R. & HOBBS, N. T. (Eds.) *Fragmentation in Semi-Arid Landscapes: Consequences for Human and Natural Systems.* Springer.

REID, R. S., NKEDIANYE, D., SAID, M. Y., KAELO, D., NESELLE, M., MAKUI, O., ONETU, L., KIRUSWA, S., KAMUARO, N. O., KRISTJANSON, P., OGUTU, J. O., BURNSILVER, S. B., GOLDMAN, M. J., BOONE, R. B., GALVIN, K. A., DICKSON, N. M. & CLARK, W. C. (2009) Evolution of models to support community and policy action with science: Balancing pastoral livelihoods and wildlife conservation in savannas of East Africa. *Proceedings of the National Academy of Sciences*.

REID, R. S., RAINY, M., OGUTU, J. O., KRUSKA, R., KIMANI, K., NYABENGE, M., MCCARTNEY, M., KSHATRIYA, M., WORDEN, J., NG'ANG'A, L., OWUOR, J., KINOTI, J., NJUGUNA, E., WILSON, C. J. & LAMPREY, R. H. (2003) People, Wildlife and Livestock in the Mara Ecosystem: the Mara Count 2002. Nairobi, Kenya, ILRI (International Livestock Research Institute).

REID, R. S., THORNTON, P. K. & KRUSKA, R. (2004) Loss and fragmentation of habitat for pastoral people and wildlife in east Africa: concepts and issues. *African Journal of Range & Forage Science,* 21**,** 171-181.

REPUBLIC OF KENYA (2006) NATIONAL TOURISM POLICY - Final Draft. Nairobi, Ministry of Tourism and Wildlife (MoTW).

REPUBLIC OF KENYA (2007) Draft Wildlife Policy, 2007. Nairobi, Ministry of Tourism and Wildlife (MoTW).
REPUBLIC OF KENYA (2009) The Wildlife (Conservation and Management) Bill 2009. Kenya: Ministry of Forestry and Wildlife (MF&W).
REPUBLIC OF KENYA (2011a) Draft WILDLIFE POLICY, 8 July 2011. Nairobi, Kenya, Ministry of Forestry and Wildlife (MoF&W).
REPUBLIC OF KENYA (2011b) The Wildlife Bill, 2011. Nairobi, Ministry of Wildlife and Forestry.
REPUBLIC OF KENYA (2012a) Sessional Paper No. 8 of 2012 on the National Policy for the Sustainable Development of Northern Kenya and other Arid Lands: "Releasing Our Full Potential". Nairobi, Ministry of State for Development of Northern Kenya and Other Arid Lands, Office of the Prime Minister.
REPUBLIC OF KENYA (2012b) Vision 2030 Development Strategy for Northern Kenya and Other Arid Lands. Nairobi, Ministry of State for Development of Northern Kenya and Other Arid Lands, Office of the Prime Minister.
RITSMA, N., HUMMEL, J. & GYELTSHEN, P. (2010) Tourism and Payments for Environmental Services: The Outlook for a Stronger Business Case to Develop Rural Tourism in Bhutan. *Mountain Forum Bulletin, January 2010.* Mountain Forum Secretariat.
ROBINSON, L. W. & BERKES, F. (2011) Multi-level participation for building adaptive capacity: Formal agency-community interactions in northern Kenya. *Global Environmental Change,* 21**,** 1185-1194.
RODRIGUEZ, L. C. (2008) A Global Perspective on the Total Economic Value of Pastoralism: global synthesis report based on six country valuations. Nairobi, Kenya, WISP (World Initaitive for Sustainable Pastoralism).
RODRIGUEZ, L. C., HENSON, D., HERRERO, M., NKEDIANYE, D. & REID, R. S. (2011a) Private farmers' compensation and viability of protected areas: the case of Nairobi National Park and Kitengela dispersal corridor. *International Journal of Sustainable Development & World Ecology***,** 1-10.
RODRIGUEZ, L. C., PASCUAL, U., MURADIAN, R., PAZMINO, N. & WHITTEN, S. (2011b) Towards a unified scheme for environmental and social protection: Learning from PES and CCT experiences in developing countries. *Ecological Economics,* 70**,** 2163-2174.
ROE, D. (2008) The origins and evolution of the conservation-poverty debate: a review of key literature, events and policy processes. *Oryx,* 42**,** 491-503.
ROE, D. (2010) Whither biodiversity in development? The integration of biodiversity in international and national poverty reduction policy. *Biodiversity,* 11**,** 13-18.
ROE, D., THOMAS, D., SMITH, J., WALPOLE, M. J. & ELLIOT, J. (2011) Biodiversity and Poverty: Ten Frequently Asked Questions - Ten Policy Implications. *IIED Gatekeeper Series No. 150.* London, International Institute for Environment and Development (IIED).
ROE, D., WALPOLE, M. & ELLIOT, J. (2010) (I) Linking biodiversity conservation and poverty reduction: What, Where and How? *Biodiversity,* 11**,** 107-124.
ROUGET, M., COWLING, R. M., LOMBARD, A. T., KNIGHT, A. T. & GRAHAM, I. H. K. (2006) Designing large-scale conservation corridors for pattern and process. *Conservation Biology,* 20**,** 549-561.
RUDNAI, J. (1979) Ecology of lions in Nairobi National Park and the adjoining Kitengela Conservation Unit in Kenya. *African Journal of Ecology,* 17**,** 85-95.
RUSSELL, D. (2009) Reliable inference for the Gini-index. *Journal of Econometrics,* 150**,** 30-40.

RUTTEN, M. M. E. M. (1992) *Selling wealth to buy poverty : the process of the individualization of landownership among the Maasai pastoralists of Kajiado District, Kenya, 1890-1990,* Saarbrücken ; Fort Lauderdale, Verlag breitenbach Publishers.
RUTTEN, M. M. E. M. (2004) Partnerships in Community-Based Ecotourism Projects: Experiences from Maasai Region, Kenya. *ACS Working Paper.* Leiden, Netherlands.
SACHEDINA, H. & NELSON, F. (2012) The Development of Payments for Ecosystem Services as a Community-based Conservation Strategy in East Africa. IN INGRAM, J. C., DECLERCK, F. & DEL RIO, C. R. (Eds.) *Integrating Ecology and Poverty Reduction: The Application of Ecology in Development Solutions.* Springer.
SACHS, J. D. (2005) *The End of Poverty: Economic possibilities of our time,* London, Penguin Books.
SACHS, J. D., BAILLIE, J. E. M., SUTHERLAND, W. J., ARMSWORTH, P. R., ASH, N., BEDDINGTON, J., BLACKBURN, T. M., COLLEN, B., GARDINER, B., GASTON, K. J., GODFRAY, H. C. J., GREEN, R. E., HARVEY, P. H., HOUSE, B., KNAPP, S., KUMPEL, N. F., MACDONALD, D. W., MACE, G. M., MALLET, J., MATTHEWS, A., MAY, R. M., PETCHEY, O., PURVIS, A., ROE, D., SAFI, K., TURNER, K., WALPOLE, M., WATSON, R. & JONES, K. E. (2009) Biodiversity Conservation and the Millennium Development Goals. *Science,* 325**,** 1502-1503.
SAFRIEL, U. & ADEEL, Z. (2005) Dryland systems. IN HASSAN, R., SCHOLES, R. J. & ASH, N. (Eds.) *Ecosystems and Human Well-being: Current State and Trends, Volume 1.* Washington, D.C., Island Press.
SAGOFF, M. (2011) The quantification and valuation of ecosystem services. *Ecological Economics,* 70**,** 497-502.
SAID, M. Y., OGUTU, J. O., KIFUGO, S., NKEDIANYE, D., REID, R. S., GICHOHI, H., SERNEELS, S., DE LEEUW, J. & LAMBIN, E. H. (Submitted) Land fragmentation in the Athi-Kaputie Plains of Kenya through fencing: implications for pastoralism and wildlife conservation.
SALZMAN, J. (2005) The promise and perils of payments for ecosystem services. *International Journal of Innovation and Sustainable Development,* 1**,** 5-20.
SANDERSON, S. E. (2005) Poverty and Conservation: The New Century's "Peasant Question?". *World Development,* 33**,** 323.
SANJAYAN, M. A. & CROOKS, R. K. (2005) Maintaining Connectivity in Urbanizing Landscapes. IN JOHNSON, A. E. & KLEMENS, W. M. (Eds.) *Nature in Fragments: The Legacy of Sprawl.* New York, Columbia University Press.
SCHMIDT-SOLTAU, K. & BROCKINGTON, D. (2007) Protected Areas and Resettlement: What Scope for Voluntary Relocation? *World Development,* 35**,** 2182-2202.
SCHNEIDER, H. (1990) Development and the Pastoralists of East Africa. IN GALATY, J. G. & SALZMAN, C. (Eds.) *Nomads in A Changing World.* Naples, Istituto Universitario Orientale.
SCHOLTE, P. (2011) Towards understanding large mammal population declines in Africa's protected areas: A west-central African perspective. *Tropical Conservation Science,* 1**,** 1-11.
SCOONES, I. (1991) Wetlands in Drylands: Key Resources for Agricultural and Pastoral Production in Africa. *Ambio,* 20**,** 366-371.
SECRETARIAT OF THE CONVENTION ON BIOLOGICAL DIVERSITY (2009) Connecting Biodiversity and Climate Change Mitigation and Adaptation: Report of the Second Ad Hoc Technical Expert Group on Biodiversity and Climate Change. *Technical Series No. 41.* Montreal, Canada, SCBD (Secretariat of the Convention on Biological Diversity).

SECRETARIAT OF THE CONVENTION ON BIOLOGICAL DIVERSITY (2010a) Global Biodiversity Outlook 3. Montreal, SCBD (Secretariat of the Convention on Biological Diversity).
SECRETARIAT OF THE CONVENTION ON BIOLOGICAL DIVERSITY (2010b) Linking Biodiveristy Conservation and Poverty Alleviation: A State of Knowledge Review. *CBD Technical Series No. 55.* Montreal, Secretariat of the Convention on Biological Diversity.
SEN, A. K. (1999) *Development As Freedom,* New York, USA, Knopf.
SENO, S. K. & SHAW, W. W. (2002) Land tenure policies, Maasai traditions, and wildlife conservation in Kenya. *Society & Natural Resources,* 15**,** 79-88.
SERNEELS, S. & LAMBIN, E. F. (2001) Impact of land-use changes on the wildebeest migration in the northern part of the Serengeti-Mara ecosystem. *Journal of Biogeography,* 28**,** 391-407.
SERNEELS, S., SAID, M. Y. & LAMBIN, E. F. (2001) Land cover changes around a major east African wildlife reserve: the Mara Ecosystem (Kenya). *International Journal of Remote Sensing,* 22**,** 3397 - 3420.
SHAW, E. & MACKINNON, H. (Eds.) (2013) *Africa Rising: A Continent's Future Through the Eyes of Emerging Scholars,* Waterloo, ON, The Africa Initiative and the Center for International Governance Innovation.
SHELLEY, B. G. (2011) What should we call instruments commonly known as payments for environmental services? A review of the literature and a proposal. *Annals of the New York Academy of Sciences,* 1219**,** 209-225.
SIKAND, J. (2007) 'Must win, shall win'. *SWARA.* Nairobi, EAWS (East African Wildlife Society).
SIKOYO, M. G., ASHLEY, C. & ELLIOT, J. (2001) The Impact of Wildlife-Based Enterprises on Local Livelihoods and Conservation in Kenya. Nairobi, Kenya, Africa Wildlife Foundation (AWF).
SILVESTRI, S., OSANO, P., DE LEEUW, J., HERRERO, M., ERICKSEN, P., KARIUKI, J., NJUKI, J., BEDELIAN, C. & NOTENBAERT, A. (2012) Greening livestock: Assessing the potential of payment for environmental services in livestock inclusive agricultural production systems in developing countries. Nairobi, International Livestock Research Institute (ILRI).
SINDIGA, I. (1995) Wildlife-based Tourism in Kenya: Land use conflicts and government compensation policies over protected areas. *Journal of Tourism Studies,* 6**,** 45-55.
SKONHOFT, A. & SOLSTAD, J. T. (1998) Investing in Wildlife: Can Wildlife Pays Its Way? *Journal of African Economies,* 7**,** 237-262.
SMIT, B. & PILIFOSOVA, O. (2001) Adaptation to climate change in the context of sustainable development and equity. IN MCCARTHY, J. (Ed.) *Climate Change 2001: Impacts, Adaptation and Vulnerability. IPCC Working Group II.* Cambridge, UK, Cambridge University Press.
SMITH, R. J. (1981) Resolving the Tragedy of the Commons by Creating Private Property Rights in Wildlife. *CATO Journal,* 1**,** 439-468.
SORLIE, E. B. (2008) The Establishment of Wildlife Conservancies in the Mara: The Case of Olare Orok Conservancy. *Department of Political Science.* Oslo, Norway, Masters Thesis, University of Oslo.
SOUTHGATE, C. R. J. (2006) Ecotourism in Kenya: The Vulnerability of Communities. *Journal of Ecotourism,* 5**,** 80-96.
SWALLOW, B. M. & BROMLEY, D. W. (1995) Institutions, governance and incentives in common property regimes for African rangelands. *Environmental and Resource Economics,* 6**,** 99-118.

SWALLOW, B. M., KALLESOE, F. M., IFTIKHAR, A. U., NOORDJWICK, M. V., BRACER, C., SCHERR, S. J., RAJU, K. V., POATS, V. S., DURAIAPPAH, A. K., OCHIENG, B., MALLEE, H. & RUMLEY, R. (2009) Compensation and Rewards for Environmental Services in the Developing World: Framing Pan-Tropical Analysis and Comparison. *Ecology and Society,* 14**,** 26.

SWALLOW, B. M. & MEINZEN-DICK, R. (2009) Payment for Environmental Services: Interactions with Property Rights and Collective Action. IN BECKMANN, V. & PADMANABHAN, M. (Eds.) *Institutions and Sustainability: Political Economy of Agriculture and the Environment - Essays in Honour of Konrad Hagerdon.* Springer.

SWART, J. A. A., FERRARO, P. J. & KISS, A. (2003) Will Direct Payments Help Biodiversity? *Science,* 299**,** 1981b-1982.

TACHE, B. & SJAASTAD, E. (2010) Pastoralists' Conceptions of Poverty: An Analysis of Traditional and Conventional Indicators from Borana, Ethiopia. *World Development,* 38**,** 1168-1178.

TALBOT, L. & OLINDO, P. (1990) The Maasai Mara and Amboseli Reserves. IN KISS, A. (Ed.) *Living with Wildlife: Wildlife Resource Management with Local Participation in Africa.* Washington, D.C., USA, World Bank.

TALLIS, H., KAREIVA, P., MARVIER, M. & CHANG, A. (2008) An ecosystem services framework to support both practical conservation and economic development. *Proceedings of the National Academy of Sciences of the United States of America,* 105**,** 9457.

TEEB (2009) The Economics of Ecosystems and Biodiversity for National and International Policy Makers.

TEEB (2010) *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations,* London and Washington, Earthscan.

TEKELENBURG, A., TEN BRINK, B. J. E. & WITMER, M. C. H. (2009) How do biodiversity and poverty relate? - An explorative study. Bilthoven, The Netherlands, The Netherlands Environmental Assessment Agency (PBL).

THOMPSON, D. M. (2005) Valuing land use options in the Maasai Mara. Draft Report of the 2004 Socio-Economic Survey "Better Policy and Management Options for Pastoral Lands". Nairobi, International Livestock Research Institute (ILRI).

THOMPSON, D. M. & HOMEWOOD, K. (2002) Entrepreneurs, elites, and exclusion in maasailand: Trends in wildlife conservation and pastoralist development. *Human Ecology,* 30**,** 107-138.

THOMPSON, D. M., SERNEELS, S., KAELO, D. & TRENCH, P. C. (2009) Maasai Mara - Land Privatisation and Wildlife Decline: Can Conservation Pay Its Way? IN HOMEWOOD, K., KRISTJANSON, P. & TRENCH, P. (Eds.) *Staying Maasai? Livelihood, Conservation and Development in East African Rangelands.* Springer.

THORNTON, P. K., JONES, P. G., OWIYO, T. M., KRUSKA, R. L., HERRERO, M., KRISTJANSON, P., NOTENBAERT, A., BEKELE, N. & OMOLO, A. (2006) Mapping Climate Vulnerability and Poverty in Africa. Nairobi, International Livestock Research Institute (ILRI).

THORNTON, P. K., VAN DE STEEG, J., NOTENBAERT, A. & HERRERO, M. (2009) The impacts of climate change on livestock and livestock systems in developing countries: A review of what we know and what we need to know. *Agricultural Systems,* 101**,** 113-127.

TIMMER, V. & JUMA, C. (2005) TAKING ROOT: Biodiversity Conservation and Poverty Reduction Come Together in the Tropics. *Environment,* 47**,** 24.

TISDELL, C. A. (2004) Economic Incentives to Conserve Wildlife on Private Lands: Analysis and Policy. *The Environmentalist,* 24**,** 153-163.

TOWNSEND, P. (2006) What is Poverty? An historical perspective. United Nations Development Programme (UNDP), International Poverty Center.
TSCHAKERT, P. (2007) Environmental services and poverty reduction: Options for smallholders in the Sahel. *Agricultural Systems,* 94**,** 75-86.
TURNER, W. R., BRANDON, K., BROOKS, T. M., GASCON, C., GIBBS, H. K., LAWRENCE, K. S., MITTERMEIER, R. A. & SELIG, E. R. (2012) Global Biodiversity Conservation and the Alleviation of Poverty. *Bioscience,* 62**,** 85-92.
TURPIE, J. K., MARAIS, C. & BLIGNAUT, J. N. (2008) The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. *Ecological Economics*, 65**,** 788-798.
UCHIDA, E., ROZELLEE, S. & XU, J. (2009) Conservation Payments, Liquidity Constraints, and Off-Farm Labor: Impact of the Grain-for-Green Program on Rural Households in China. *American Journal of Agricultural Economics,* 91**,** 70-86.
UCHIDA, E., XU, J., XU, Z. & ROZELLE, S. (2007) Are the poor benefiting from China's land conservation program? *Environment and Development Economics,* 12**,** 593-620.
UNCCD/UNDP/UNEP (2009) Climate Change in the African Drylands: Options and Opportunities for Adaptation and Mitigation. UNCCD (United Nations Convention to Combat Desertification)/UNDP (United Nations Development Programme)/UNEP (United Nations Environment Programme).
UNDP (1997) Human Development Report 1997: Human Development to Eradicate Poverty. New York, UNDP (United Nations Development Programme).
UNEP-WCMC (2007) Biodiversity and Poverty Reduction: the importance of biodiversity for ecosystem services. Cambridge, UK, United Nations Environment Programme-World Conservation Monitoring Center (UNEP-WCMC).
UNEP (2006) Africa Environment Outlook 2. Nairobi, UNEP (United Nations Environment Programme).
UNEP (2009) Kenya: Atlas of Our Changing Environment. Nairobi, UNEP (United Nations Environment Programme).
UNRUH, J. D. (2008) Carbon sequestration in Africa: The land tenure problem. *Global Environmental Change,* 18**,** 700-707.
UPTON, M. (2004) The Role of Livestock in Economic Development and Poverty Reduction. *PPLPI (Pro-Poor Livestock Policy Initiative) Working Paper No. 10.* Rome, FAO (Food and Agricultural Organisation of the United Nations).
URA, K., STRINGER, R. & BULTE, E. H. (2009) Managing Willdife Damage to Agriculture in Bhutan: Conflicts, Costs and Compromise. IN LIPPER, L., SAKUYAMA, T., STRINGER, R. & ZILBERMAN, D. (Eds.) *Payment for Environmental Services in Agricultural Landscapes: Economic Policies and Poverty Reduction in Developing Countries.* Food and Agriculture Organisation of the United Nations (FAO) and Springer.
VALENTINE, P. & BIRTLES, A. (2004) Widlife Watching. IN HIGGINBOTTOM, K. (Ed.) *Wildlife Tourism: Impacts, Management and Planning.* Altona, Victoria, Australia, Common Ground Publishing Pty Ltd in association with Cooperative Research Center for Sustainable Tourism.
VAN DE SAND, I. (2012) Payments for Ecosystem Services in the Context of Adaptation to Climate Change. *Ecology and Society*, 17**,** 11-24.
VAN NOORDWIJK, M., LEIMONA, B., JINDAL, R., VILLAMOR, G. B., VARDHAN, M., NAMIREMBE, S., CATACUTAN, D., KERR, J., MINANG, P. A. & TOMICH, T. P. (2012) Payments for Environmental Services: Evolution Toward Efficient and Fair Incentives for Multifunctional Landscapes. *Annual Review of Environment and Resources,* 37**,** 389-420.

VATN, A. (2010) An institutional analysis of payments for environmental services. *Ecological Economics,* 69**,** 1245-1252.
VICTURINE, R. & CURTIN, C. (2010) Financial Incentives for Rangeland Conservation: Addressing the 'Show-Us-the-Money' Challenge. IN DU TOIT, J. T., KOCK, R. & DEUTSCH, J. C. (Eds.) *Wild Rangelands: Conserving Wildlife While Maintaining Livestock in Semi-Arid Ecosystems.* Blackwell Publishing Ltd.
VYAS, S. & KUMARANAYAKE, L. (2006) Constructing socio-economic status indices: how to use principal components analysis. *Health Policy and Planning,* 21**,** 459-468.
WAITHAKA, J. (2002) The Role of Community Wildlife-based Enterprises in Reducing Human Vulnerability and Environmental Degradation: The Case of Il Ngwesi Ecotourism Project, Kenya. Nairobi, Kenya, Africa Conservation Center (ACC).
WALPOLE, M. J., KARANJA, G., SITATI, N. & LEADER-WILLIAMS, N. (2003) Wildlife and People: Conflict and Conservation in Masai Mara, Kenya. *IIED Wildlife and Development Series no 14.* London, UK, International Institute for Environment and Development (IIED).
WALPOLE, M. J. & LEADER-WILLIAMS, N. (2001) Masai Mara tourism reveals partnership benefits. *Nature,* 413**,** 771-771.
WATKINS, K. & ALEMAYEHU, W. (2012) Financing for a Fairer, More Prosperous Kenya: A Review of the Public Spending Challenges and Options for Selected Arid and Semi-Arid Counties. *Center for Universal Education Working Paper 6.* Washington, D.C., Brookings Institution.
WCED (1987) *Our Common Future,* Oxford, Oxford University Press (OUP) for the World Commission on Environment and Development (WCED).
WEF (2011) The Travel &Tourism Competetiveness Report 2011: Beyond the Downturn. Geneva, Switzerland, The World Economic Forum (WEF).
WELLS, M., BRANDON, K. & HANNAH, L. (1992) People and Parks: Linking Protected Area Management with Local Communities. Washington D.C., World Bank.
WELLS, M. & MCSHANE, T. O. (2004) Integrating protected area management with local needs and aspirations. *Ambio,* 33**,** 513-519.
WENDLAND, K. J., HONZÁK, M., PORTELA, R., VITALE, B., RUBINOFF, S. & RANDRIANARISOA, J. (2010) Targeting and implementing payments for ecosystem services: Opportunities for bundling biodiversity conservation with carbon and water services in Madagascar. *Ecological Economics,* 69**,** 2093-2107.
WERTZ-KANOUNNIKOFF, S., LOCATELLI, B., WUNDER, S. & BROCKHAUS, M. (2011) Ecosystem-based adaptation to climate change: What scope for payments for environmental services? *climate and development,* 3**,** 143-158.
WEST GATE CONSERVANCY (2012) West Gate Financial Sustainability Plan. Said Business School, University of Oxford.
WESTERN, D. (2010) The Worst Drought: Tipping or Turning Point? *SWARA.* East Africa Wildlife Society (EAWS).
WESTERN, D. & MANZOLILLO NIGHTINGALE, D. A. (2003) Environmental Change and the Vulnerability of Pastoralists to Drought: A Case Study of the Maasai in Amboseli, Kenya. *Africa Environment Outlook: Past, Present and Future.* UNEP (United Nations Environment Programme).
WESTERN, D., RUSSELL, S. & CUTHILL, I. (2009) The Status of Wildlife in Protected Areas Compared to Non-Protected Areas in Kenya. *PloS ONE,* 4**,** 1-5.
WESTERN, D., RUSSELL, S. & MUTU, K. (2006) The status of wildlife in Kenya's protected and non-protected areas. *First Stakeholders Symposium of the Wildlife Policy And Legislation Review, Nairobi (Kenya) 27-28 September, 2006.* Nairobi, Kenya, African Conservation Center (ACC).

WESTERN, D. & WRIGHT, R. M. (Eds.) (1994) *Natural Connections,* Island Press.
WILKIE, D. S., CARPENTER, J. F. & ZHANG, Q. (2001) The under-financing of protected areas in the Congo Basin: so many parks and so little willingness-to-pay. *Biodiversity and Conservation,* 10**,** 691-709.
WILLIAMS, A. P. & FUNK, C. (2010) A westward extension of the tropical Pacific warm pool leads to March through June drying in Kenya and Ethiopia. *USGS Open-File Report 2010-1199.* US Geological Survey.
WONG, K., MARLENE, R. & DURAIAPPAH, A. K. (2005) Connecting poverty and ecosystem services: focus on Kenya. UNEP (United Nations Environment Programme) and IISD (the International Institute for Sustainable Development).
WORLD BANK (2001) World Development Report 2000/2001: Attacking Poverty. Washington DC.
WORLD BANK (2005) *How Much is an Ecosystem Worth?,* Washington, D.C., World Bank.
WORLD BANK (2008) Kenya Wildlife Conservation Leasing Demonstration Project. World Bank.
WORLD BANK (2009) Convenient Solutions to an Inconvenient Truth: Ecosystem-based Approaches to Climate Change. Washington, DC., The World Bank.
WORLD BANK (2011) Kenya's Tourism: Polishing the Jewel. The World Bank.
WORLD RESOURCES INSTITUTE (2005) World Resources 2005: The Wealth of the Poor-Managing Ecosystems to Fight Poverty. Washington, D.C., World Resources Institute, UNEP and World Bank.
WORLD RESOURCES INSTITUTE (2007) Natures Benefits in Kenya: An Atlas of Ecosystems and Human Well-Being. Washinton DC and Nairobi, World Resources Institute.
WTTC (2012) Travel & Tourism Economic Impact 2012: Sub-Saharan Africa. London, UK, World Travel & Tourism Council (WTTC).
WÜNSCHER, T., ENGEL, S. & WUNDER, S. (2008) Spatial targeting of payments for environmental services: A tool for boosting conservation benefits. *Ecological Economics,* 65**,** 822-833.
WUNDER, S. (2005) Payments for environmental services: some nuts and bolts. Jakarta, Indonesia, Center for International Forestry Research (CIFOR).
WUNDER, S. (2007) The Efficiency of Payments for Environmental Services in Tropical Conservation. *Conservation Biology,* 21**,** 48-58.
WUNDER, S. (2008) Payments for environmental services and the poor: concepts and preliminary evidence. *Environment and Development Economics,* 13**,** 279-297.
WUNDER, S. & ALBAN, M. (2008) Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. *Ecological Economics,* 65**,** 685-698.
WUNDER, S., ENGEL, S. & PAGIOLA, S. (2008) Taking stock: A comparative analysis of payments for environmental services programs in developed and developing countries. *Ecological Economics,* 65**,** 834-852.
WUNDER, S. & WERTZ-KANOUNNIKOFF, S. (2009) Payments for Ecosystem Services: A New Way of Conserving Biodiversity in Forests. *Journal of Sustainable Forestry,* 28**,** 576 - 596.
WWF (2009) Guide to Conservation Finance. Washington, DC, WWF (USA).
YOUNG, P. T. (2006) Declining Rural Populations and the Future of Biodiversity: Missing the Forest for the Trees. *Journal of International Wildlife Law and Policy,* 9**,** 319-334.
ZABEL, A. & ENGEL, S. (2010) Performance payments: A new strategy to conserve large carnivores in the tropics? *Ecological Economics,* 70**,** 405-412.
ZBINDEN, S. & LEE, D. R. (2005) Paying for Environmental Services: An Analysis of Participation in Costa Rica's PSA Program. *World Development,* 33**,** 255-272.

ZILBERMAN, D., LIPPER, L. & MCCARTHY, N. (2008) When could payments for environmental services benefit the poor? *Environment and Development Economics,* 13**,** 255-278.

ZWAAGSTRA, L., SHARIF, Z., WAMBILE, A., DE LEEUW, J., SAID, M. Y., JOHNSON, N., NJUKI, J., ERICKSEN, P. & HERRERO, M. (2010) An Assessment of the response to the 2008-2009 drought in Kenya. A report to the European Union Delegation to the Republic of Kenya. Nairobi, ILRI (International Livestock Research Institute).

APPENDICES

APPENDIX I –A: Inventory of Community Wildlife Conservancies and Wildlife Payment for Ecosystem Services (PES) Schemes in Kenyan Arid and Semi-Arid Lands (ASAL)

Version: February, 2012

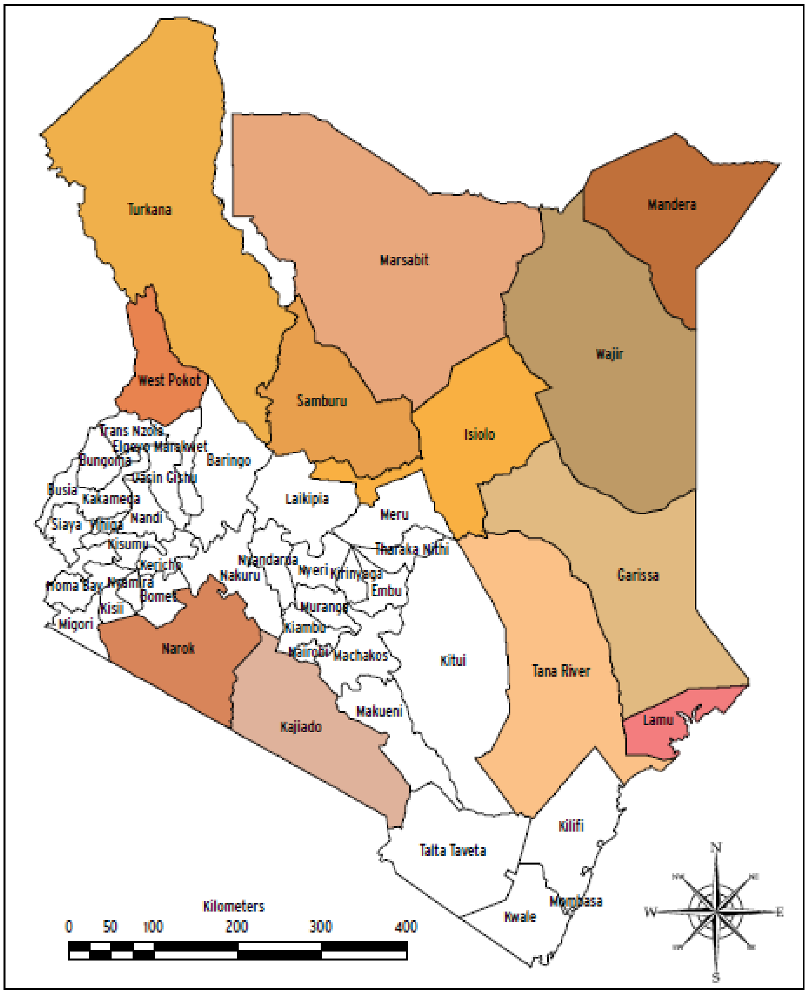
District	Name	Production System (Total Area)	Funding typology	No. landowners/ members	Land tenure category	Year of Registration	Land Area (Ha) (Include core)	Core conservation area (Ha)	Payment rate (US$/ha/yr)
Narok	Enoonkishu	wildlife + livestock	Market	66	Private	2010	6,566	6,566	
Narok	Ol Choro Oiroua	wildlife + livestock	Market	96	Private	1992	6,879	6,879	
Narok	Mara North Conservancy	wildlife + livestock	Market	800	Private	2009	30,955	30,955	36
Narok	Lemek	wildlife + livestock	Market	180	Private	2002	6,860	6,860	
Narok	Motorogi	wildlife + livestock	Market	106	Private	2007	5,466	5,466	36
Narok	Olare Orok Conservancy	Wildlife	Market	157	Private	2006	9,720	9,720	43
Narok	Naboisho	wildlife + livestock	Program	506	Private	2010	20,946	20,946	29
Narok	Ol Kinyei	wildlife	Market	70	Private	2006	4,856	4,856	19
Kajiado	Kimana	wildlife + livestock	Market	843	Group Ranch	1996	25,120	6,000	
Kajiado	Imbirikani	wildlife + livestock	Market	4585	Group Ranch	1996	129,895	10,000	
Kajiado	Elerai	wildlife + livestock	Market	1	Private	2007	122,875	5,000	
Kajiado	Esenlenkei	wildlife	Market	3000	Group Ranch	1997	74,794	4,300	
Kajiado	Kuku Motikanju	wildlife + livestock	Unknown	6200	Group Ranch		96,000	7,000	
Kajiado	Kitirua Conservation Area	wildlife + livestock	Market	11000	Group Ranch	1999	147,013	12,140	
Kajiado	Shompole	wildlife + livestock	Market	1404	Group Ranch	1999	62,869	14,000	4
Kajiado	Olkiramatian	wildlife + livestock	Market		Group Ranch	2004	21,612	10,000	
Kajiado	Ilengarunyoni	Wildlife + livestock	Program		Group Ranch	2008			
Kajiado	Mailua	Wildlife + livestock	Program	1200	Group Ranch	2008	110,000		
Kajiado	Kilotome	Wildlife + livestock	Program	85	Private	2008	5,000		6
Kajiado	Osupuko	Wildlife + livestock	Program	50	Private	2008	3,000		6

Kajiado	Rombo Emampuli	Wildlife + livestock	Market		Group Ranch	2008	38,365		
Kajiado	Laro	Wildlife	Market	1	Private	2004		26,000	
Kajiado	Olerai	wildlife + livestock	Program	26	Private	2007	3,560		
Kajiado	Kitengela	wildlife + livestock	Program	260	Private	2001	16,000		11
Kwale	Golini-Mwaluganje	wildlife	Program	127	Private	1994		3,600	
Laikipia	Lekurruki	wildlife + livestock	Program	3700	Group Ranch	1999	7,000	1,584	
Laikipia	Il Ngwesi	wildlife + livestock	Program	6000	Group Ranch	1995	8,675	6,677	
Laikipia	Naibunga	wildlife + livestock	Program	12000	Group Ranch	2001	43,000	5,600	
Laikipia	Koija Conservancy	wildlife + livestock	Program	1500	Group Ranch	2002	710	200	
Laikipia	Il Polei Conservancy	wildlife	Program		Group Ranch		17,401	2,240	
Samburu	Keno				Group Ranch			69,354	
Samburu	Kalama	wildlife + livestock	Program	2000	Group Ranch	2002	95,000	3,150	
Samburu	West Gate	wildlife + livestock	Program	3500	Group Ranch	2004	40,350	887	
Samburu	Meibae	wildlife + livestock	Program	10000	Group Ranch	2006	87,611		
Samburu	Ngirgir	wildlife + livestock	Program		Group Ranch			49,644	
Samburu	Namunyak Wildlife Trust	wildlife + livestock	Program	8000	Group Ranch	1995	64,987	2,000	
Samburu	Sera Conservancy	wildlife + livestock	Program	8000	Trust land	2001	300,000	33,325	
Samburu	Biliqo Gulesa	wildlife + livestock		5000	Trust land	2007			
Garissa	Ishaqbini Community Conservancy	Wildlife + livestock	Program	6000	Trust land	2007	720		
Baringo	Ruko Conservancy	Wildlife + livestock	Program	1500	Trust land	2006	19,000	7,689	

APPENDIX I –B: The list of the Arid and semi-arid Districts and their respective Counties
Data source; KNBS (2009; 2010; 2011)

The Arid Counties	**The Semi-Arid Counties**
(1) Tana River (incl. Tana Delta) (2) Turkana (incl. Turkana Central, North and South) (3) Marsabit (incl. Chalbi and Laisamis) (4) Mandera (incl. Mandera Central, East and West) (5) Wajir (incl. Wajir South, West, North and East) (6) Samburu (incl. Samburu North, Central and East) (7) Isiolo (incl. Garbatulla) (8) Garissa (incl. Ijara, Fafi and Lagdera) (9) West Pokot (incl. Pokot North, Central and East) (10) Baringo (incl. Baringo North and Koibatek)	(1) Lamu (2) Kilifi (3) Kwale (4) Taita-Taveta (incl. Taita and Taveta) (5) Kitui (incl. Mwingi) (6) Kajiado (inc. Kajido North, Central and Loitokitok) (7) Narok (incl. Narok South, North and Trans-Mara) (8) Laikipia (incl. Laikipia North, East and West)

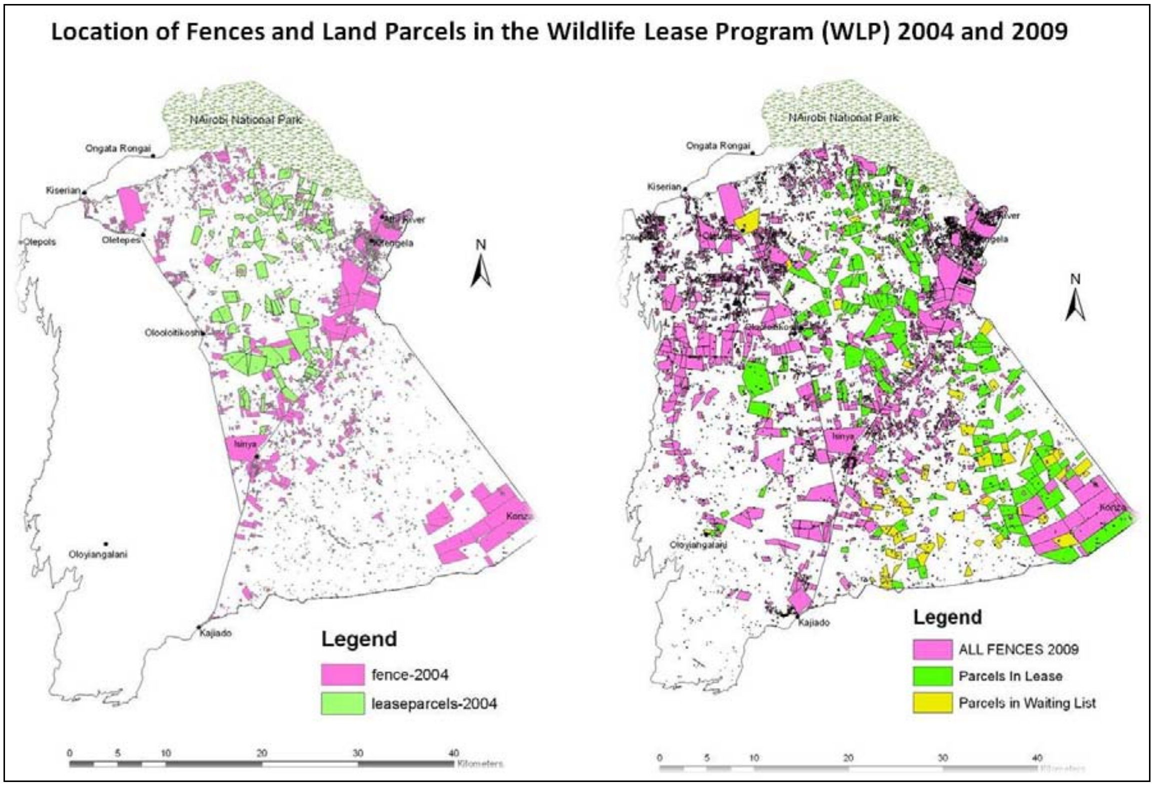
APPENDIX I –C: Kenya's New County Map with selected ASAL Counties
Source; Watkins and Alemayehu (2012)



Source: created by The World Bank: Nairobi (2012)

APPENDIX II

APPENDIX II –A: Map of Athi-Kaputie study site showing the land parcels enrolled in the Wildlife Lease Program (WLP), parcels targeted for enrolment (waiting list) and the distribution of fences in (a) 2004 and (b) 2009
Source; ILRI



APPENDIX II –B: Results of a correlation analysis of income, land, shoats (goats and sheep), cattle, and livestock for the households surveyed in Athi-Kaputiei Plains (n=164).

	Income2009	Land (ha)	Shoats (TLU)	Cattle (TLU)	Livestock (TLU)
Income2009	1.000				
Land	0.152	1.000			
Shoats (TLU)	0.433	0.249	1.000		
Cattle (TLU)	0.582	0.161	0.666	1.000	
Livestock (TLU)	0.579	0.203	0.828	0.969	1.000

APPENDIX II –C: Results of the diagnostic of model specification error using the link test

(Pseudo-r^2 = 0.1774)

WLP Status	Coefficient	SE	z	P>|z|	LCL	UCL
Hat	1.121478	0.2340907	4.79	0.000	0.6626691	1.580288
Hat square	-0.1358654	0.1139636	-1.19	0.233	-0.3592299	0.0874992
Intercept	0.1037833	0.2027386	0.51	0.609	-0.293577	0.5011437

APPENDIX II –D: Results of Hosmer-Lemeshow's goodness-of-fit test for the model

Number of observations = 158; Number of groups = 10

Hosmer-Lemeshow $\chi^2_{df=8}$ = 13.35, P = 0.1005.

		WLP Participants		Non-Participants		
Group	**Prob**	**Observation**	**Expected**	**Observation**	**Expected**	**Total**
1	0.2553	2	2.7	14	13.3	16
2	0.3358	6	4.8	10	11.2	16
3	0.3940	4	5.9	12	10.1	16
4	0.4778	4	7.0	12	9.0	16
5	0.5274	7	7.6	8	7.4	15
6	0.6013	14	8.9	2	7.1	16
7	0.6734	12	10.2	4	5.8	16
8	0.7644	12	11.6	4	4.4	16
9	0.8745	12	13.3	4	2.7	16
10	0.9899	13	14.0	2	1.0	15

APPENDIX II –E: Results of the Variance Inflation Factor (VIF) test for multi-collinearity

Variable	VIF	1/VIF
ndvi_mav4	2.12	0.471664
dist_twn_km	1.75	0.571936
cai	1.53	0.654951
dist_prsch~m	1.45	0.687528
education	1.40	0.716592
adult_lab	1.35	0.740842
inc_total09	1.30	0.772152
wldlife_pred	1.29	0.777525
curr_plot_ha	1.27	0.786612
child_depr~o	1.23	0.815708
credit_indx	1.21	0.824629
gender	1.17	0.851700
emp_status	1.13	0.884619
Mean VIF	**1.40**	