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Impact of Land Use Change and Climate Variability on Watershed Hydrology in the Mara River Basin, East Africa

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Impact of Land Use Change and Climate Variability on Watershed Hydrology in the Mara River Basin, East Africa

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Declaration of conformity

I confirm that this copy is identical with the original dissertation entitled:

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Abstract

Land use change and climate variability are the main drivers of watershed hydrological processes. The main objective of this study was to assess the impact of land use change and climate variability on hydrology of the Mara River Basin in East Africa. Land use maps generated from satellite images were analyzed using the intensity analysis approach to determine the patterns, dynamics and intensity of land use change. Changes in measured streamflow caused separately by land use change and climate variability were separated using the catchment water-energy budget based approach of Budyko framework. The information on past impact of climate variability on streamflow was used to develop a runoff sensitivity equation which was then used to predict the future impact of climate change on streamflow. Finally, the impact of agroforestry on watershed water balance was predicted using SWAT (Soil and Water Assessment Tool) model. Deforestation and expansion of agriculture were found to be dominant and intensive land use changes in the watershed. The deforestation was attributed to illegal encroachment and excision of the forest reserve. The deforested land was mainly converted to small scale agriculture particularly in the headwaters of the watershed. There was intensive conversion of rangeland to largescale mechanized agriculture which accelerated with change of land tenure (privatization). The watershed has a very dynamic land use change as depicted by swap change (simultaneous equal loss and gains of a particular land use/cover) which accounted for more than half of the overall change. This implies that reporting only net change in land use (of MRB) underestimates the total land use change. The results show that streamflow of Nyangores River (a headwater tributary of the Mara River) significantly increased over the last 50 years. Land use change (particularly deforestation) contributed 97.5% of change in streamflow while the rest of the change (2.5%) was caused by climate variability. It was predicted that climate change would cause a moderate 15% increase in streamflow in the next 50 years. SWAT model simulations suggested that implementation of agroforestry in the watershed would reduce surface runoff, mainly due expected improvement of soil infiltration. Baseflow and total water yield would also decrease while evapotranspiration would increase. The changes in baseflow (reduction) and evapotranspiration (increase) were attributed to increased water extraction from the soil and groundwater by trees in agroforestry systems. The impact of agroforestry on water balance (surface runoff, baseflow, water yield and evapotranspiration) was proportional to increase in size of the watershed simulated with agroforestry. Modelling results also suggested that climate variability within the watershed has a profound effect on the change of water balance caused by implementation of agroforestry. It is recommended that authorities should pay more attention to land use change as the main driver of change in watershed hydrology of the basin. More effort should be focused on prevention of further deforestation and agroforestry may be considered as a practical management strategy to reverse/reduce degradation on the deforested parts of the watershed currently under intensive cultivation.

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Abbreviations/acronyms

AMCOW	African Ministers’ Council on Water
APFM	Associated Programme on Flood Management
CN	Curve Number
DEM	Digital Elevation Model
EC	European Commission
ESRI	Environmental Systems Research Institute
FAO	Food and Agriculture Organization of the United Nations
GoK	Government of Kenya
GWP	Global Water Partnership
HRU	Hydrological Response Unit
IWRM	Integrated Water Resource Management
MMNR	Maasai Mara National Researve
MRB	Mara River Basin
NEMA	National Environment Management Authority
PHU	Plant Heat Unit
SCS	United States Soil Conservation Service
SWAT	Soil and Water Assessment Tool
UNEP	United Nations Environmental Programme
UN-HABITAT	United Nations Human Settlements Programme.
USEPA	United States Environment Protection Agency
WRMA	Water Resources Management Authority

1 Chapter One: Introduction

1.1 Background and motivation

Environment is an integral part of life, as human beings derive from the environment several ecosystem (environmental) services that are necessary for their survival (Nelson et al., 2009; d'Arge et al., 1997). Water is one of the basic needs that human beings cannot live without; indeed water is life! Therefore, water-related (hydrological) ecosystem services provided by the environment (e.g. provision, regulation and purification of freshwater) are quite valuable and important for human well-being (Francesconi et al., 2016; Terrado et al., 2014; Nedkov and Burkhard, 2012; Pert et al., 2010). This underscores the importance of sound watershed management for continued provision of hydrological ecosystem services (Fan et al., 2016; Daily et al., 2009; Brauman et al., 2007). From a hydrological point of view, a watershed includes all land contributing water (surface and ground water) to a reference point (outlet). It is therefore obvious that land comprising of any watershed would generally be under other uses such as forests, agriculture and urban centers, which might commonly be considered 'primary' land uses. This means that watersheds provide (and are expected to) other important ecosystem services (e.g. food production), besides provision of hydrological ecosystem services (Power, 2010; Nelson et al., 2009). In some cases, enhanced provision of some ecosystem services may also lead to reduced capacity of watersheds to provide other services (Jin et al., 2015; Bennett et al., 2009) e.g. intensive cultivation to maximize food production may also lead to increased soil erosion and consequently degradation of water quality (Butler et al., 2013). Watershed degradation (e.g. through intensive cultivation) may alter soil infiltration properties which consequently affects how a watershed partitions rainwater into various components of water balance (e.g. surface runoff, lateral flow, groundwater recharge) (Recha et al., 2013; Nedkov and Burkhard, 2012). This shows how human activities on land (watershed) affect the availability and quality of water resources (Crossman et al., 2013). This linkage between water, land and people make it necessary to widen the scope of watershed management beyond the 'water resources' (Butler et al., 2013; Mwangi, 2013; Falkenmark and Rockström, 2004). Integrated water resource management (IWRM) is a broader and effective approach of watershed management that is now accepted globally. IWRM is a process that promotes coordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems (GWP, 2000). As such, IWRM advocates for integration of management of land and water because of the interdependence between the two resources.

Indeed, land use change is one of the main drivers of change in watershed hydrology (Fan et al., 2016; Xu et al., 2014; Jin et al., 2015; Tomer and Schilling, 2009). The growing world population raises the demand for some basic needs in life such as food and housing. To meet these demands, land transformations (e.g. expansion and intensification of agriculture and growth of urban centers) have occurred in many parts of the world (Hosonum et al., 2012; Mundia and Aniya, 2006). Deforestation, expansion of agriculture and growth of urban centers are some of the most common and widespread land use/cover changes worldwide (Mubea et al., 2014; Hosonum et al., 2012; Geist and Lambin, 2001). Parts of many indigenous forests (e.g. Amazon and Congo Basin) have been cleared and converted to agriculture and settlements for the expanding local population (Ernst et al., 2013; Morton et al., 2006; Pfaff, 1991). For example, Gibbs et al. (2010) estimated that over 50% of new agricultural land in the tropics between 1900 and 2000 came from intact forests and DeFries et al. (2010) identified urban population growth and agricultural trade as the main drivers of forest loss in the tropics between 2000 and 2005. The mechanism of land use/cover changes at local, regional and global levels is complex and requires deeper understanding of the processes and driving forces of land use change (Hosonum et al., 2012; Rudel et al., 2009; Geist and Lambin, 2001). Assessment of land use change over time helps in understanding the land use change processes and linking the changes to possible driving forces (Teixeria et al., 2014). This information is necessary for development of environmental management and conservation measures, prediction of future land use and modelling the effect of past and future land use change on different components of natural systems such as watershed hydrology or biodiversity (Ku, 2016; de Chazal and Rousevell, 2009). Observation of changes in land use and land cover over different scales is now possible through the use of satellite data. In the past, information of land use/cover could only be obtained from small areas through ground surveys or aerial photographs. Satellite data has now extended the coverage of ground observation to larger areas and the frequency of observations is also high making it possible to assess land cover change in larger watersheds at different intervals of time (DeFries and Eshleman, 2004; Al-doski et al., 2013).

Understanding the consequences of land use/cover change on hydrological processes is of major interest to hydrologists and water resources managers (Jin et al., 2015; Mao and Cherkauer, 2009). Whereas some direct consequences such as water demands due to changes in land use practices (e.g. irrigation and urbanization) may be easily estimated, estimating changes in water balance and quality resulting from altered hydrological processes of infiltration, groundwater recharge and evapotranspiration is not straight-forward; it requires a thorough understanding of the complex watershed hydrology (DeFries and Eshleman, 2004). This is further complicated by the fact that any observed change in streamflow (which is the main component of water balance measured in many watersheds, globally) also includes the impact of climate variability (Tomer and Schilling, 2009; Wang and Hejazi, 2011). Climate variability is another main driving force of change of watershed hydrology (Fan et al., 2016; Terrado et al., 2014; Ye et al., 2013). Quantification of the impacts on streamflow caused separately by land use change and climate variability improves understanding of hydrological processes for better management of watersheds (Wang, 2014). This understanding is, for example, helpful to water managers who may be interested to know which of the two drivers has contributed more change to streamflow. The information is useful in designing effective watershed management interventions and strategies (Fan et al., 2016). Hydrologists may also be interested with this information as a basis for modelling future impacts of either land use change or climate change on watershed hydrology for planning purposes (Tomer and Schilling, 2009; DeFries and Eshleman, 2004). Selection of feasible and practical watershed management strategies/interventions is an important task of watershed management planning (Mwangi et al., 2015a; Giri and Nejadhashem, 2014). Some management interventions are however long-term (e.g. afforestation and agroforestry) and their effect on watershed hydrology are hard to reverse (Mwangi et al., 2016a; Zhang and Zhang, 2011; Zhang et al., 2008). Therefore, use of hydrological models to predict the impacts of such interventions on watershed hydrology before they are implemented, aids the planners in decision making (Fan et al., 2016). The impact of climate change on water resources is also a concern for watershed management (Hirabayashi et al., 2013; Dai, 2013). At a local level, the impact may be increase in magnitude and frequency of floods or it could be increase in severity of droughts which may affect water supply (Apurv et al., 2015; Hirabayashi et al., 2013; Lott et al., 2013; Dai, 2013). Thus, prediction of the impacts of climate change on water resources at a local level enables water managers in designing appropriate mitigation measures.

The main aim of this study was to assess the impact of land use change and climate variability on hydrology of the Mara River Basin in East Africa. Mara River Basin (MRB) (Figure 1.1) is a transboundary watershed shared between Kenya and Tanzania. In Kenya, the watershed cuts across three semi-autonomous counties (Nakuru, Bomet and Narok). Watershed management of the basin therefore requires integration of water-related interests of the three counties in Kenya as well as those for Tanzania on the downstream end of the basin. Previous studies in the watershed have reported that land use change (especially deforestation and expansion of agriculture) has caused change in watershed hydrology. The Government of Kenya (GoK, 2009) has also been concerned with the extent of deforestation in the Mau Forest complex, one of the main 'water towers' of the country and which is a source of several rivers including the Mara. Agricultural cultivation is currently predominant in the formerly deforested areas and the Government of Kenya is keen on restoring forest cover in as much area as possible.

Although previous studies in the MRB have reported that land use change has caused change in hydrology, none of the studies analyzed the measured streamflow data to find out how it has changed over time. Furthermore, no study has investigated how much of observed change in streamflow is separately caused by land use change and climate variability. A previous land use change study by Mati et al. (2008) focused on net changes in land use. However, net changes in land use/cover may not reveal the whole extent of land use change because loss of a particular land use category at one point of the watershed may be accompanied by a gain of similar size of the same land use category at another location within the watershed during the same time interval. This kind of information may not be revealed when land use change focuses on net changes, yet the change may also have an effect on some watershed characteristics e.g. soil infiltration properties. Therefore, a knowledge gap exists on the overall extent of land use change including swap changes and how the change has impacted the water yield of the watershed, separately from climate variability. It is also not clear how the increase in tree cover desired by the Government of Kenya would impact on water availability in the MRB. Information on the size of the watershed that can sustainably be put under additional tree cover is also lacking.

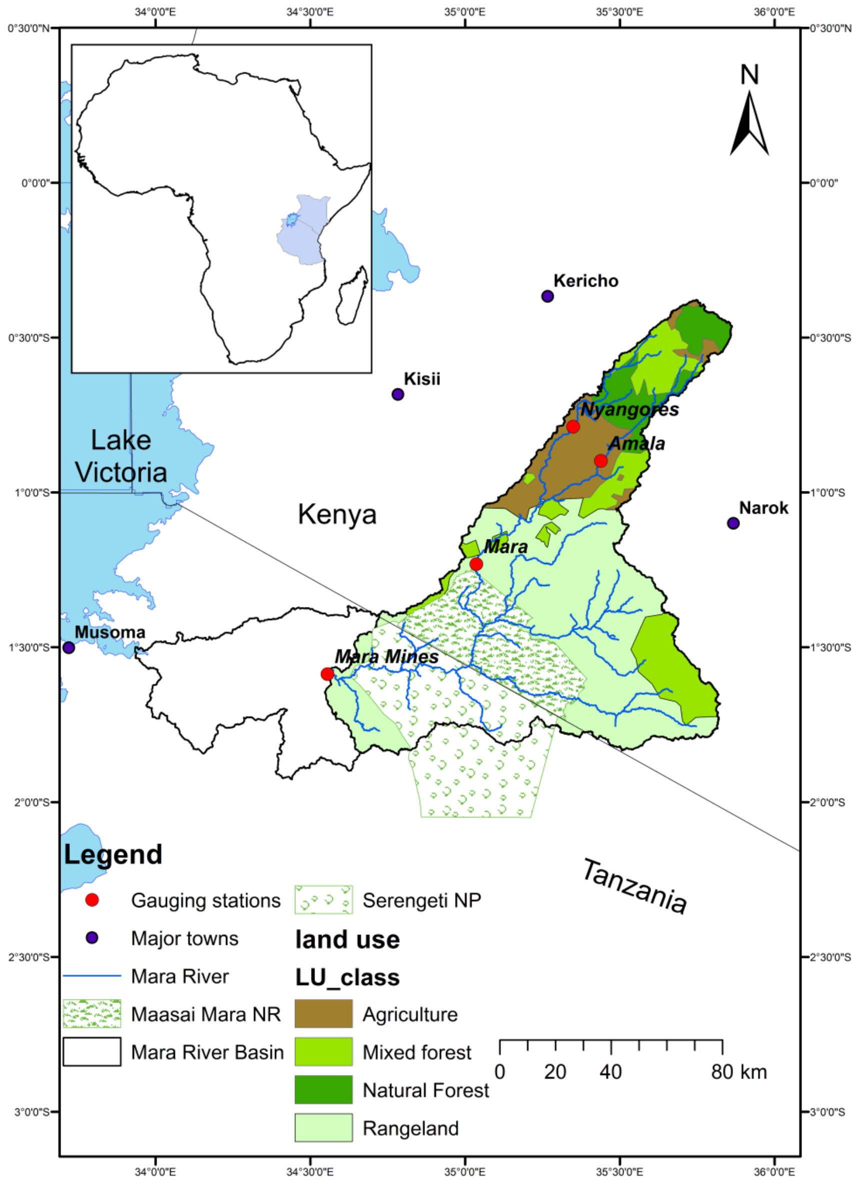

Figure 1.1: Mara River Basin with land use distribution and location of gauging stations.

1.2 Research questions

To fill these knowledge gaps, this study aimed to answer three sets of research questions organized as follows:

1) How has land use in the Mara River Basin changed over the last 40-50 years? How are the dynamics of land use change particularly regarding forest and agriculture in the watershed?

2) Has recorded streamflow in the Mara River changed in the last 50 years? If yes, how much of this change has been separately caused by land use change and climate variability? How will climate change affect streamflow of the watershed in the near future?

3) What is the impact of agroforestry (as a feasible/desired future land use change) on catchment water balance? What size of the watershed can sustainably be put under tree cover?

1.3 Structure of the Dissertation

The dissertation is organized in six chapters; four of which consist of a compilation of research papers. Two papers have already been published in peer-reviewed journals while the third has been submitted to a peer-reviewed journal. The forth paper has already been published as a book chapter. Each of the three sets of the research questions is answered in a separate chapter/paper.

Chapter one gives general introduction and an overview of the study. It highlights the background, the problem, the objectives and the research questions addressed by the study. **Chapter two** introduces and presents a review of the state-of-the-art in watershed management. The concept of integrated water resources management (IWRM) as an effective approach of watershed management is highlighted. Watershed management planning which is a fundamental process of watershed management is presented. The chapter places much emphasis on watershed assessment which is a key component of watershed planning. A case study of watershed assessment in Sasumua watershed, Kenya is presented. In the context of Mara River basin, the bulk of the work of this study as presented in chapters 3, 4 and 5 is part of watershed assessment, which is the foundation of watershed planning.

Chapter 3 attempts to answer the first set of research questions. Satellite imagery of the MRB between 1976 and 2014 are analyzed in four consecutive intervals by intensity analysis approach (Aldwaik and Pontius, 2012) to assess and quantify land use changes that have occurred in the basin. Overall land use change which includes swap change, in addition to net change, is assessed to give a better overview of the entire land use change. The chapter also focuses on transitions between land use categories to identify dominant land use transitions with a particular focus on deforestation and expansion of agriculture in the watershed.

Chapter 4 deals with the second set of research questions. Recorded streamflow data was analyzed to find out whether there has been any change. Observed change in streamflow of Nyangores River, one of the upper tributaries of Mara River, was separated using the catchment water-energy budget approach of Budyko framework (Budyko, 1974; Roderick and Farquhar, 2011) to estimate how much of the change was caused separately by land use change and climate variability. The impact of climate change on streamflow of the Mara River in the near future (i.e. in the next 20 and 50 years) was also predicted.

In **chapter 5**, the impact of agroforestry on catchment water balance was assessed (third set of research questions). Agroforestry is one of the feasible watershed management strategies that has been proposed to increase the forest cover in the already deforested (and converted to agriculture) parts of the Mau forest (Government of Kenya, 2009). It is considered as a practical solution to aid in recovery of some of the degraded parts of the watershed currently under intensive cultivation. The findings of this chapter are helpful for selection of management strategies/interventions during the initial or subsequent cycles of the adaptive watershed management planning processes discussed in chapter 2. The SWAT (Soil and Water Assessment Tool) model (Arnold et al., 1998) was used to assess the impact of agroforestry on surface runoff, baseflow, evapotranspiration and groundwater recharge and overall water yield of the MRB. SWAT is a physically based model widely used for prediction of the impact of land management on water sediment and agricultural chemical yields (Gassman et al., 2007, 2010). It is capable of studying ecosystem processes in a systematic manner and hence able to evaluate ecosystem services, particularly the hydrological ecosystem services (Francesconi et al., 2016).

Finally, **chapter 6** gives a summary and a synthesis of the main findings from preceding chapters. The general conclusions of the study are based on the results of chapters 3, 4 and 5. General recommendations of the study particularly regarding watershed management and conservation are also presented.

1.4 List of publications

1. Mwangi HM, Julich S, Patil SD, McDonald MA, Feger KH. 2016. Relative contribution of land use change and climate variability on discharge of upper Mara River, Kenya. Journal of Hydrology: Regional studies 5, 244-260.

2. Mwangi HM, Julich S, Patil SD, McDonald MA, Feger KH. 2016. Modelling the impact of agroforestry on hydrology of Mara River Basin in East Africa. Hydrological Processes, 30, 3139-3155.

3. Mwangi HM, Julich S, Feger KH. 2015. Introduction to Watershed Management. In *Tropical Forestry Handbook*, 2nd edition, Pancel L, Köhl M (eds). Springer-Verlag: Berlin, Heidelberg. DOI: 10.1007/978-3-642-41554-8_153-1.

4. Mwangi HM, Lariu P, Julich S, Patil SD, McDonald MA, Feger KH. 2016. Land use change intensity of Mara River Basin, East Africa. Applied Geography (*submitted*)

2 Chapter two: Introduction to Watershed Management

Publication (this chapter has been published in springer as a book chapter)

Mwangi HM, Julich S, Feger KH. 2015. Introduction to Watershed Management. In *Tropical Forestry Handbook, 2nd Ed.,* Pancel L, Köhl M (eds). Springer-Verlag: Berlin, Heidelberg. DOI 10.1007/978-3-642-41554-8_153-1.

Abstract

Scarcity and threats to freshwater resources from pollution, climate change, and overexploitation have made it increasingly important to have sound watershed management. The link between land, water, and people has further made it necessary to widen the scope of watershed management beyond the "water resources." Overall ecosystem functions as well as the improvement of socioeconomic status of the local communities are of paramount importance for the success of watershed management. The chapter provides a general overview of watershed management and modern challenges originating from climate change and land-use pressures. It highlights some of the critical issues that should be addressed for successful watershed management with a regional emphasis on tropical Africa. In this context, sustainable forest management and also agroforestry is a key factor in water resources management in general and upland resources development in particular. Integrated water resources management (IWRM) including stakeholder participation, livelihood improvement, flood risk management, and financing of watershed management is presented. Furthermore, the scheme of watershed planning process which is fundamental for the development and implementation of watershed management plans is stressed. Watershed assessment, a key component of watershed planning, is outlined based on a case study in the Sasumua dam watershed, Kenya.

2.1 Introduction

Availability of freshwater resources is essential for human life and well-being, economic development, and ecosystem health (Falkenmark and Rockström, 2004). Both terrestrial and freshwater aquatic ecosystems require freshwater to thrive for continued supply of ecosystem services to human beings. The human society equally requires freshwater for its survival and economic development. Therefore, both the ecosystems and the human society are linked through the freshwater cycle (Figure 2.1) which is a part of the entire hydrological cycle.

Human demands for water are usually broken down into five major water use sectors (WWAP, 2012):

- Food and agriculture (mostly irrigation), which accounts for about 70 % of water withdrawals globally
- Energy
- Industry
- Human settlements, which includes water for drinking and household uses such as cooking, cleaning, hygiene, and some aspects of sanitation
- Ecosystems (both aquatic and terrestrial), whose water demands are determined by the water required to sustain or restore the benefits to people (ecosystem services)

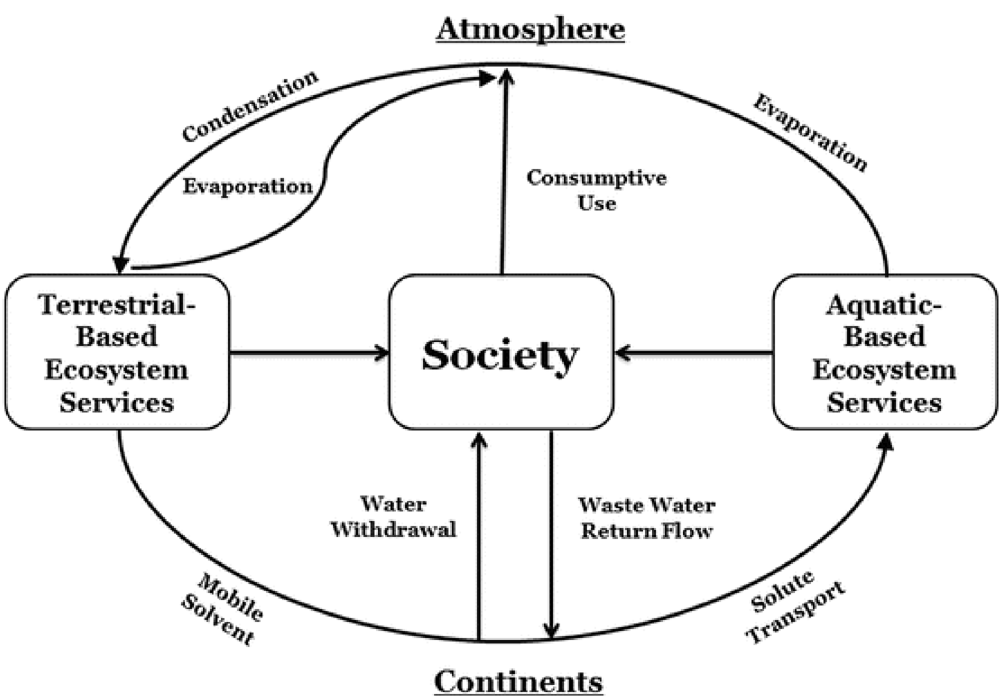

Figure 2.1: Linkages between freshwater cycles, human livelihood, and ecosystems (Adapted from Falkenmark and Rockström, 2004)

These uses, which are all beneficial for human well-being, compete for the available freshwater. This competition compounded by the uneven distribution of water resources over time and space and the way human activity is affecting that distribution are the underlying causes of water crises in many parts of the world (e.g., Vörösmarty, 2009). Furthermore, climate change is superimposed on the complex water cycling in watersheds. Notably, the increase of extreme events like drought and heavy rainfall puts additional pressure on water supplies. There is increasing concern related to population growth, overutilization of groundwater aquifers, waterlogging and salinization, pollution through urban and industrial wastes, fertilizers and pesticides from agricultural land, and flooding of cultivated, urban, and industrial areas. Many of these problems are related to changes in land use, i.e., deforestation and other reduction of close-to-nature vegetation forms like wetlands, urbanization, and intensification of agricultural production (UNEP, 2009).

To address the various water crises, innovative ways of enhancing water security are required. Water security is defined as the availability of an acceptable quantity and quality of water for health, livelihoods, ecosystems, and production, coupled with an acceptable level of water-related risks to people, environments, and economies (WWAP, 2012). This includes the sustainable use and protection of water systems, protection against water-related hazards (i.e., floods and droughts), sustainable development of water resources, and safeguarding water functions and services for humans and the environment. Managing the challenges of water security therefore requires an integrated management approach based on sound understanding of the watershed processes and interactions among watershed components, i.e., land, water, and people. An integration of natural and social science-based research is important to improve that understanding.

The 1992 United Nations Conference on Environment and Development (UNCED) held in Rio de Janeiro emphasized, in Chapter 18 of its Agenda 21, the holistic management of freshwater as a finite and vulnerable resource. The chapter advocates for water resources planning and management for the protection of the quality and supply of freshwater resources and proposes application of integrated approaches to the development, management, and use of water resources. This integrated approach, known as integrated water resources management (IWRM), is now being adopted globally. Ten years after UNCED, a major impetus to improving IWRM was provided at the Johannesburg 2002 World Summit on Sustainable Development (WSSD). A large number of countries agreed to the Johannesburg Plan of Implementation, calling for the development and implementation of IWRM and water efficiency strategies, plans, and programs at national and at regional levels. The first step in IWRM process (Figure 2.2) is to create an enabling environment by changing policies and laws and creating new (or rearrange) institutions that have a legal mandate for water resources management. With right policies, legislations, and institutions, IWRM planning and implementation become faster and smoother.

For watershed management to be successful, the focus should go beyond the "water resource" itself and include socioeconomic and environmental concerns. Development activities in the watershed should be incorporated in watershed management plans, and there should be a concerted effort to improve livelihood. Thus, understanding the dynamics and the structure of the local communities is important. The community should be actively involved in watershed management because its success highly depends on whether or not they embrace the watershed management efforts. The watershed management programs should also be "environmental conscious," i.e., seeking to preserve and protect terrestrial and aquatic biodiversity, preventing land degradation, and avoiding/reducing unsustainable land-use practices. Indeed, IWRM not only advocates for sustainable development and management of land, water, biomass, and other resources for human well-being but also the protection of natural ecosystems. In this context, sustainable forest management and agroforestry is a key factor in water resources management in general and upland resources development in particular. Forests provide a wide range of environmental services, some of which are water related (i.e., protection from soil erosion, optimal water retention, and minimal leaching of nutrients and contaminants). Thus, conservation of headwater forest catchments (notably tropical cloud forests; see Julich et al., 2015) is particularly important for sustainable provision of watershed services.

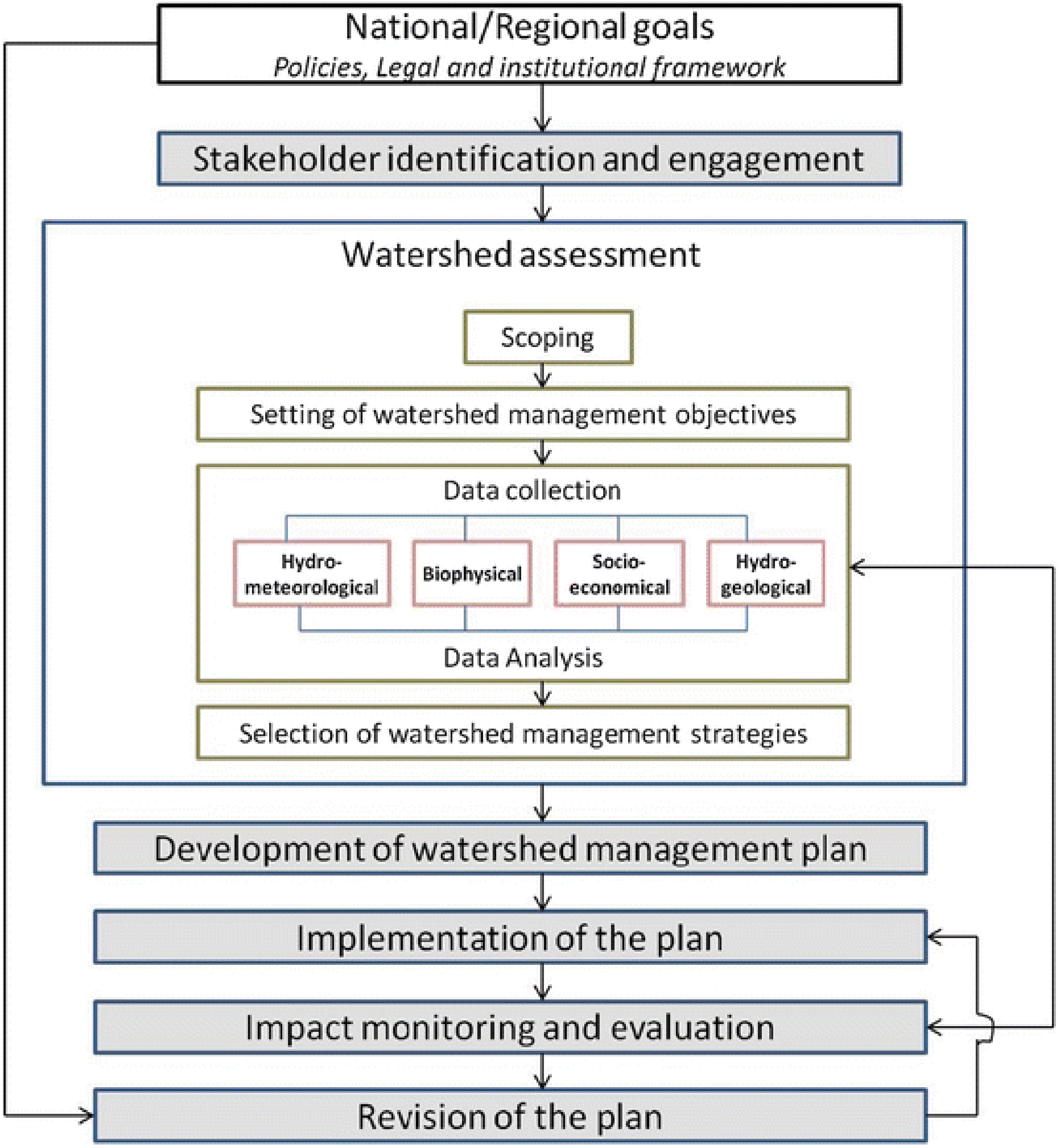

Figure 2.2: Watershed management planning process.

The complex relationship between land and water (including the life they support) necessitates a drainage-based watershed management approach. This approach brings all water users and potential water polluters within a particular watershed on a platform where they can share the water equitably for development and also control its pollution. The upstream and downstream water-related interests are commonly addressed and managed which minimizes water-related conflicts. The scale of individual watershed management units is an issue that is crucial for meaningful participation of stakeholders in the watershed. In general, the scale of watershed units should be large enough to include the major upstream and downstream interests and small enough to ensure active participation of all stakeholders and allow comprehensive watershed assessment. In trans-boundary water basins, collaboration among the countries or states sharing the water basin is required. To do this, creation of international river basin organization with representation of member countries or states is required. Such organizations ensure that interests of member countries are addressed. Examples of international basin organizations in sub-Saharan Africa, Europe, and Asia are summarized in Table 2.1.

Table 2.1: Examples of international basin commissions in Africa, Europe, and Asia.

Organization	Basin/watershed	Participating countries
Lake Victoria Basin Commission (LVBC)	Lake Victoria	Kenya, Tanzania, Uganda
Nile Basin Initiative (NBI)	Nile River	Burundi, DR Congo, Egypt, Kenya, Rwanda, South Sudan, Sudan, Sudan, Tanzania
Okavango River Basin Commission (OKACOM)	Okavango River	Angola, Botswana, Namibia
Zambezi Watercourse Commission (ZAMCOM)	Zambezi River	Angola, Botswana, Malawi, Mozambique, Namibia, Tanzania, Zimbabwe, Zambia
Mekong River Commission (MRC)	Mekong	Cambodia, Lao PDR, Thailand, Vietnam
International Commission for the Protection of the Rhine (ICPR)	Rhine	Switzerland, France, Germany, Luxemburg, the Netherlands
International Commission for the protection of the Danube River (ICPDR)	Danube	Austria, Bosnia and Herzegovina, Bulgaria, Croatia, Czech Republic, Germany, Hungary, Moldova, Montenegro, Romania, Slovakia, Slovenia, Serbia, Ukraine
International Commission for the Protection of the Elbe River (ICPER)	Elbe	Germany, Czech Republic

2.2 Integrated Water Resources Management (IWRM)

Integrated Water Resources Management (IWRM) recognizes that water resources have many dimensions. Thus, the objective of IWRM is to integrate all sectors which utilize or are related to water resources including the different institutions and policies for efficient management of water resources. The Global Water Partnership (GWP), an international network founded in 1996 to foster IWRM, defines IWRM as “a process which promotes the coordinated development and management of water, land, and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystem” (GWP, 2000).

The IWRM principles focus on a holistic multi-sectoral approach in water management which integrates/includes governance, institutional, scientific, technical, socioeconomic, and environmental aspects of water management (UNEP, 2010). The four principles are:

1. Freshwater is a finite and vulnerable resource, essential to sustain life, development, and the environment.
2. Water development and management should be based on a participatory approach, involving users, planners, and policy makers at all levels.
3. Women play a central part in the provision, management, and safeguarding of water.
4. Water has an economic value in all its competing uses and should be recognized as an economic good.

The important question to ask in IWRM then is: “integrate what and why?”
Sound water resources management must deal with the natural and the socioeconomic components of the watershed. Natural and human systems should therefore be integrated for efficient and sustainable management of the water resources. Integration should be done both within and between the systems. Integration is intended to change the traditional fragmented and uncoordinated development and management of water resources (GWP, 2000).

Under natural system, the focus should be on:

- Integration of land and water management: Water (of good quality and in sufficient quantities) is essential for most land developments (e.g., irrigation, industrial or domestic water supply). To ensure sustainable supply of good quality freshwater, good management of land (terrestrial ecosystems) is required. Thus, land and water resources are interdependent and require integrated management approach.
- Integration of “blue” and “green” water management: Efficiency of water use is crucial in managing the rising demand and competition for water among various uses. Efficient use of green water (water used for biomass production, i.e., soil water used or “lost” in the process of evapotranspiration) would save “blue” water (freshwater in lakes, river, springs, and groundwater beyond the rooting zone) (cf. Julich et al., 2015). For example, increasing irrigation efficiency will save water for other uses, notably for domestic or industrial use.

- Integration of surface and groundwater management: Surface and groundwater resources are connected through the hydrological cycle which also affects their availability in space and time.
- Integration of management of water quality and quantity: Water pollution is one of the major threats facing dwindling freshwater resources. Pollution impairs water quality and makes it unsuitable for most uses and therefore adds more pressure on the remaining freshwater resources.
- Integration of upstream and downstream water-related interests: Excessive upstream water use could lead to insufficient water for downstream uses. Equitable sharing of water is required for sustainable development and to avoid water-related conflicts. Upstream human activities should assure availability and good quality of water for downstream users at all times. Land use in upstream areas should maintain natural flood retention and minimize erosion losses.
- Integration of freshwater management and coastal zone management: Management of freshwater should consider the needs of coastal zones for water quantity and quality.

Human system integration involves:

- Cross-sectoral policy development integration: Water is a core pillar in development. Water is required in different sectors such as domestic, agricultural, industrial, and environmental. All these sectors traditionally have separate policies. Development of such policies should consider the specific water requirement and availability as well as the respective impact on water quality.
- Integration of stakeholders in watershed planning and decision-making: Stakeholder participation ensures that all interests and concerns of various stakeholders are taken care of in the watershed management planning.
- Integrating water and wastewater management: To minimize the pollution of freshwater resources by wastewater and ease pressure on freshwater resources, wastewater reuse and recycling is required. Opportunities for wastewater reuse and recycling are available for other water uses that do not require strict water quality standards (relative to drinking water standards), e.g., irrigation, gardening, and process water cooling (UNEP, 2005).

The whole idea of IWRM is therefore to facilitate efficient and smooth water resources management for sustainable development. Adoption of IWRM at national level helps in faster and smoother watershed management planning and implementation at the local (watershed and sub-watershed) level. Most countries are making good progress in planning and implementation as agreed in the Johannesburg 2002 World Summit on Sustainable Development (UNEP, 2012). A worldwide survey carried out by UN Water showed that about 80 % of the countries are at advanced stages of changing their water policies and law to accommodate IWRM, while 65 % have developed IWRM plans out of which 34 % are implementing the plans (UNEP, 2012). In Africa, a recent study commissioned by African Minister's Council on Water found that 76 % of the countries are in the process of implementing national laws to allow an enabling environment for IWRM, while 44 % have already developed and are implementing national plans (AMCOW, 2012).

2.3 Participatory Watershed Management

In participatory watershed management, stakeholders in the water resources development, conservation, and management are actively involved from the start of decision-making. In this case, decisions are not imposed on them, but they are part of the decision-making process where they can share their views, concerns, interests, and fears and also offer their resources in terms of time, finances, skills, and knowledge to the watershed management process. In the past, both top-down and bottom-up approaches in watershed management failed because of lack of support by stakeholders who did not feel ownership of the watershed management decisions (Johnson et al. 2002). People often resist decisions imposed on them if they were not part of the decision-making process. Since the modern scope of watershed management goes beyond the water resources itself and includes improving livelihoods and sustainable developments within the watershed, the net of stakeholders is wide. It includes individuals, groups, and institutions that have direct interest in the water resources, e.g.:

- Water users
- People whose actions are likely to impair water quality, e.g., smallholder farmers and industries
- People in forest management or with interest in environmental conservation
- People with scientific knowledge (notably local ecology) of the watershed

- Government and nongovernmental institutions with interest or mandate in natural resource management and livelihood improvement

Participation goes beyond informing them of the watershed management programs. It involves collecting and incorporating their views, fears, and interests in the watershed management plans. With the wide range of stakeholders with a variety of resources, participation includes collaborating with them for the benefit of the watershed management (FAO, 2006).

2.4 Livelihoods and Watershed Management

People and all their activities are an integral part of watersheds. The social, religious, economic, and political aspects of life should be considered in watershed management planning as well since they determine the level of success of watershed management efforts. Those watershed management efforts that are contrary to community traditions, beliefs, norms, and values are likely to fail. Livelihood comprises of capabilities, resources, and activities required in order to live (Chambers and Conway, 1991). It generally consists of everything tangible or non-tangible that people rely on to make a living. People use the resources at their disposal to make a living. The resources can be natural/physical (e.g., land, water, crops, forests, animals, etc.), financial (e.g., income, savings), human (education, skills, knowledge, etc.), and social (interactions, traditions, beliefs, etc.) (FAO, 2006). Actually it is peoples' everyday business to make a living and continuously improve their living standard. Therefore, watershed management plans should be established within the livelihood framework. Watershed management cannot be independent of livelihoods, and watershed management should put as much emphasis on ways to improve livelihood as it does on ways to prevent watershed degradation. To improve livelihoods, special efforts have to be made to optimize the use of the resources available to local communities.

Understanding the local communities' way of living is crucial before designing watershed management programs. For instance, simply providing a toilet for a community that practices open defecation may not be enough way for improving their sanitation condition. People may shun the use of the toilet because of their beliefs. During watershed management planning, understanding local livelihoods may help to identify the resources available at household level and develop sustainable strategies to optimize their use. It would further help to design sustainable solutions to the existing environmental risks that are also acceptable to the local community.

2.5 The Role of Forest Management in Watershed Management

Forests natural or managed are essential elements of the landscape and provide valuable ecosystem services to the communities in the watershed like soil protection, carbon sequestration, and production of timber, firewood, fruits, and fodder. Forests play an important role in the hydrological cycle of a watershed. Compared to other land uses like agriculture and urban areas, some hydrological processes are dominant and determine the partitioning of rainfall into streamflow and evapotranspiration (fluxes of "blue" and "green" water (Julich et al., 2015). For example, evapotranspiration rates in forests are higher than in agricultural systems due to higher canopy interception and a deeper rooting system which can access soil water from deeper horizons (Calder, 2005). Changes in forest cover can lead to changes in magnitude and dynamics of water yield, increase of dry season weather flow, and higher sediment load from soil erosion. The nature and magnitude of changes in the water cycle and related watershed response depend on the watershed characteristics (i.e., soils and relief/topography), type of change in forest cover (selective logging vs. conversion of forest into agricultural land), as well as the overall distribution and dynamics of land use in the watershed (Recha et al., 2012; Bruijnzeel, 2004; Julich et al., 2015). For example, the conversion of forest to agriculture in one part of the watershed and the simultaneous abandonment of agriculture areas in another part could balance the hydrological impacts at the catchment scale. It has been widely accepted that forests are crucial to the sustainable management of water ecosystems and resources (Calder, 2002, 2005; FAO, 2013). Therefore, watershed management should also promote appropriate forest management and protection in order to maximize the positive effects of forests or other tree-based vegetation structures on water resources (water management through forest management).

Finally, it has to be recognized that the hydrological function of forests is an important but just one in a whole bundle of benefits which forests provide to society. There is a range of productive, conservation, amenity, environmental, and livelihood benefits. Therefore, a key challenge faced by land, forest, and water managers is to maximize this wide range of multi-sectoral forest benefits without detriment to water resources and ecosystem function.

2.6 Flood Risk Management

Floodplains are preferred for human settlement and socioeconomic development because of their proximity to rivers, guaranteeing rich soils for agriculture, abundant water supplies, means of transport, and aesthetic purposes (APFM, 2007). This essentially increases flood risk because the magnitude of flood disaster is not a factor of flood water alone but is also augmented by the vulnerability of the people living in the floodplains and the economic activities present. In big cities of some developing countries, informal settlements spring up in public land such as riparian areas along the water courses (e.g., Mukuru and Mathare slums in Nairobi, Kenya). Informal settlements (slums) are characterized by congestion, unplanned structures made of weak building materials, and high levels of poverty – all of which constitute high vulnerability of the inhabitants to flood disaster.

Traditionally, flood defence was the main focus for flood protection where structural solutions such as dykes were seen as the ultimate solutions (Chang et al., 2010). Loss of lives, displacement of people, and destruction of property occur when the structural flood protection measures fail. Such catastrophic flooding events have demonstrated the limitations of flood defence and the need for more strategic, holistic, and long-term approaches to manage floods in the form of flood risk management (Khatibi 2011; Johnson and Priest, 2008). Therefore, flood risk management should integrate strategies for flood protection before the flood (e.g., dykes, early warning systems), managing the flood disaster during a flood event (e.g., evacuation) and post-flood recovery measures. The strategies should comprise of structural and non-structural measures applicable to specific locations within the watershed. Appropriate strategies should be bundled in flood risk management plans which should be developed as part of IWRM (APFM, 2007). In Europe, for example, the EU member states are required by the Flood Risk Directive (EC, 2007) to carry out flood risk assessment and develop flood hazard and flood risk maps as well as to establish flood risk management plans for areas with significant flood risk (Mostert and Junier, 2009).

Preventive nonstructural flood risk management strategies, where appropriate, may include retaining as much water in the watershed/landscape as possible to reduce flood peaks. Land-use changes that minimize water infiltration (e.g., urbanization and deforestation) contribute to the increase of flash floods. The presence of undisturbed forests in the watershed helps to retain considerable amount of water through enhanced infiltration, consequently reducing the peaks of flash floods (Wahren et al., 2012; Recha et al., 2012). Watershed characteristics such as watershed slope and soil properties, e.g., depth, porosity, water storage capacity, organic matter content, and antecedent soil moisture condition, determine the extent to which forests reduce the peaks and volume of floods. The impact of forest (or any other land use) on flood peaks tends to be higher in smaller watersheds than in large basins where variability in rainfall characteristics such as areal distribution and intensity may override the effects of land use (Bruijnzeel, 2004; FAO, 2013).

It is important to realize that the flood protection provided by forest in regions where floods are generated (notably in the upland of catchments) has inherent limitations, particularly related to the magnitude of storm events (cf. Calder et al., 2007; Bathurst et al., 2011). One major limitation depends on the site-specific retention potential, notably in the soil. Nevertheless, it is reasonable to expect that the mitigation potential of forests and tree-based vegetation structure on flood formation would become larger with a corresponding increase of the forested area of a watershed particularly in the upstream headwater areas. The “forest effect” is most significant for the more frequent small- and medium-sized floods (Wahren et al., 2012). Another limiting factor for the “forest effect” in flood mitigation may be the lack of land for afforestation given other competitive land-use requirements, notably cropland agriculture at sites which due to specific topographic and soil properties would be more effective for flood protection than others.

In order to reduce flooding risks in populated downstream areas, it may be required that rivers obtain more space to accommodate excess water during flooding events. In this context, forests in alluvial floodplain including riparian may play a crucial role. However, in many cases forest land along river courses have been reduced at the expense of settlements and agriculture. As a consequence, existing floodplain forests which are adapted to flooding events should be protected and maintained. For the restoration of disturbed systems and/or planting of new forests dedicated to water retention in floodplains, detailed knowledge is needed in terms of expected flooding dynamics (i.e., water levels, duration of flooding, sediment loads, groundwater flows), related tolerance of tree species against flooding, and potential morphological and biotic responses (cf. Bayley, 1995). The effect of forests (trees) on water availability during dry seasons or droughts needs to be considered as well. Furthermore, the acceptance of local population has to be assured by establishing participatory planning and management processes (cf. Roggeri, 1995).

2.7 Financing Watershed Management Programs

One of the challenges that face watershed management programs is the source of finances required for their implementation. Collaboration of various stakeholders allows sharing of costs and resources available from different partners. It is possible to have different organizations (e.g., NGOs) or different government institutions having similar or related projects in the same watershed. In a participatory watershed management, related projects or programs which can complement each other should collaborate and share financial and human resources. When the local communities support the watershed management programs, they can as well contribute their time, labor, and finances to undertake some activities, e.g., soil and water conservation in their farms. Government is a major stakeholder in watershed management. In many countries, government charges some levies for water abstraction. Some of this money should be channeled back for watershed management.

Recently, the concept of watershed economics has gained popularity where the concept of environmental services is used as a source of funds to finance watershed management activities (Brauman et al., 2007). The approach recognizes and appreciates the true value of environmental goods and services which have always being regarded as “free goods and services” from nature, e.g., provision of fresh quality water and carbon sequestration. In this arrangement, the beneficiaries of the environment services (whether locally, nationally, or internationally) provide some incentives to the stewards of the environmental services for the conservation of the ecosystem (cf. Mwangi et al., 2015b). This arrangement, commonly known as Payment for Ecosystem Services (PES), is a potential source of financial resources.

2.8 Watershed Management Plans

Watershed (river basin) management plans are the key management tools in IWRM. In Europe such river basin management plans are required to be established by the Water Framework Directive (EC, 2012). A watershed management plan is a time-bound strategy that describes how to achieve management objectives. The plan includes the goals, problems, feasible interventions, actions, participants and their roles, time frame, and resources required to carry out the stipulated actions for an effective watershed management.

The goals of watershed management planning may differ from one watershed to the other based on local priorities. Generally, the objectives revolve about equitable sharing of water resources, environmental protection, and enhanced economic and social development (Pegram et al., 2013). Equity in sharing of water resources should be ensured with regard to the spatial distribution of the users and also among the different uses. Water resources should thus be shared equitably at international (among countries sharing a trans-boundary basin), national (among various administrative regions), and local (sub-watershed upstream and downstream) levels. Equity is also required between different uses, e.g., domestic, agriculture, energy, industrial, etc. Environmental protection may target issues like control of pollution of water resources, biodiversity conservation, rehabilitation of degraded lands, etc. Promotion of social and economic development can be achieved through improving livelihoods of the local communities and national economy at national level.

Watershed management planning (Figure 2.2) is not a one-way process; it is iterative and adaptive in nature with cycles of a few years. The lessons learned from one cycle are incorporated in the subsequent cycles. It should also be flexible enough to allow informed modification of strategies during the implementation phase if necessary. Furthermore, watershed planning is a participatory process where all stakeholders are involved and actively participate in the process. The stakeholders may include government departments, private companies, individuals, community groups and association, scientific community and NGOs dealing with agriculture, forestry, hydropower, and environmental protection. The list may be long but should generally include all water users (of "green" or "blue" water), potential water polluters, and anyone interested in environmental-related matters and livelihood improvement. This ensures that the interests of both upstream and downstream water users are taken into account.

Watershed planning is done at two levels – short term and long term. Short-term planning is done for short period cycles (e.g., 5-year cycles), while long-term planning (for like 20 or 25 years) is a high-level strategic planning that takes into consideration the development agenda and political climate of a country or of member states for trans-boundary watersheds. Short-term watershed planning is done at the watershed and sub-watershed level and should aim at identifying and solving the specific water resources problems at the community level. The short-term watershed plans should be designed with the aim of achieving the long-term watershed plans.

The watershed management process encompasses the following steps:

1) Stakeholder identification and engagement
2) Watershed assessment
    a. Scoping
    b. Setting out watershed management objectives
    c. Data collection and analysis
    d. Selection of watershed management strategies
3) Development of watershed plan
4) Implementation of the plan
5) Monitoring and evaluation
6) Revision of the plan

Stakeholder Identification and Engagement

As already pointed out, watershed management is very wide in terms of the components/sectors it involves, e.g., agriculture, forestry, nature and biodiversity conservation, etc. This makes it necessary to first identify all the stakeholders in the watershed who use or are likely to pollute water, who may be affected by watershed management decisions, who make water resources-related decisions, and generally anyone who has an interest in the water resources management including those who can facilitate or block watershed management efforts. Possible stakeholders in a watershed may be:

- Land owners and managers
- Pastoral communities
- All water abstractors (whether individuals or private and public organizations), e.g., agricultural farms community-based water organizations, schools, hotels, water supply companies, etc.
- Government ministries or department (at national, federal, state, and county levels), e.g., water, irrigation, environment, agriculture, fisheries, forestry, etc.

- Government of countries sharing trans-boundary water resources
- Research institutions, e.g., universities, colleges, and public or private research institutions in fields related to natural resources management, e.g., in water, forestry, agriculture, etc.
- Community-based organizations
- Nongovernmental organizations
- Environmental conservation groups
- Individuals, groups, and companies whose activities are likely to impair the water quality

All these stakeholders should be reached and informed of the intended watershed planning and the need for their involvement in the process to offer ideas and also raise their concerns. Of course, there may be some challenges when reaching some of the stakeholders, and therefore it is important to show them how they are going to benefit from the whole process in short and in the long term. This stage of stakeholder engagement influences the success of watershed management because if most of the stakeholders embrace the process, they will contribute in the planning, and it will remove some hurdles that are likely to emerge at advanced stages of the planning process.

To get the maximum benefit from the stakeholders, it is prudent to know or group the stakeholders into various categories depending on their status, skills, and potential roles in the planning process (e.g., USEPA, 2008):

- Stakeholders with technical skills, e.g., researchers, scientists, and government representatives
- Stakeholders who can provide financial resources, e.g., NGOs, government ministries and department, companies, etc.
- Stakeholders who can provide local or scientific information about the watershed, e.g., village elders, universities, research institutes, etc.
- Stakeholders who could be having programs (already running or planned) that can be integrated in the watershed management planning
- Stakeholders who will have a direct role in implementing the watershed plan

- Stakeholders who may be affected by the implementation of the plan
- Any other relevant category depending on the composition of the stakeholders and their relevance in the watershed management planning and implementation

Innovative ways of engaging the stakeholders should be used so as to solicit their views, ideas, and concerns. This could be in the form of public meetings, surveys, specialized committees, consultative forums, etc. The idea is to actively involve the stakeholders in the planning process where their resources in form of skills, knowledge, ideas, finances, connections will be used to benefit the process. The concerns, views, and interests of various stakeholders are also discussed and incorporated in the plans in the best way possible for the benefit of all the stakeholders and the environment.

Watershed Assessment

A fundamental part in the process of watershed planning or integrated water resources management (IWRM) is the evaluation of the current conditions of the water and natural resources as well as socioeconomic status in the watershed. In this step, all the information required for identification of issues/problems related to water resources in the respective watershed and specific measures to address them is collected and analyzed. The more formal definition of water resources assessment is “the determination of the sources, extent, dependability and quality of water resources for their utilization and control” (WMO, 2012). The assessment of the socioeconomic conditions of the watershed is helpful in understanding the existing and possible future growth in key water-using sectors of the economy, social dynamics, and interactions and possible social impacts of watershed management decisions.

Through watershed assessment process, the following information should be obtained:

Quantity and demand of freshwater resources in the catchment

Since rainfall is the major source of freshwater resources in a watershed, it is necessary to have information about the annual precipitation amounts as well as information about its spatial and temporal (seasonality) variability (cf. Julich et al., 2015). Additionally, it is important to know about the quantity and frequency of river discharges since they form the basis for planning of water-related developments, water sharing, and flood risk management. Assessment of available freshwater resources should also include groundwater and other surface water resources, e.g., freshwater lakes and reservoirs. Another important aspect of the quantification of water resources is the assessment of the current and future water demand in the watershed for:

- Domestic use
- Industrial production
- Hydropower generation
- Irrigation

Quality of water resources

For the planning process, it is important to know about the status of the quality of the water resources as well as possible sources of pollution. In general, stream water quality can be impaired via point sources like discharges of untreated or insufficiently treated wastewater from municipalities and industries. Another source of impairment is the nonpoint source pollution from the landscape in form of agricultural chemicals like fertilizer or pesticides as well as sediments eroded from unprotected soils in the landscape (e.g., after clear-cutting or forest fires).

Socioeconomic conditions in the watershed

The socioeconomic conditions in the watershed determine future water demand and influence land use and therefore impact quantity and quality of the water resources. On the other hand, most measures developed by the watershed management planning will also have socioeconomic impacts. Thus, data and information on water use, current population, and growth rates are necessary. In order to select or design management strategies that are able to improve livelihoods, information of the sources and levels of income of the community are necessary. Information on values, norms, beliefs, and social interaction of the local communities may also be required in order to design strategies that are acceptable.

Other data and information necessary to characterize the watershed include biophysical characteristics (e.g., topography, soils, land use, hydrogeology, etc.) and climate (e.g., rainfall, temperature, evapotranspiration rates, etc. [cf. Julich et al., 2015]). The extent and quality of information required depend on the objectives and scope of the watershed management.

Watershed Assessment Process

Scoping

To get a preliminary understanding of the watershed and all the underlying issues, the first task is to carry out scoping exercise. Scoping helps to identify the scale and full extent of watershed problems, issues to be addressed, and external issues that may constrain or facilitate the process such as the government policies and legal framework. It is based on the existing data and information as well as discussion with the stakeholders. It therefore aids in setting the boundaries of the planning process and the geographical boundaries of the watershed. With scoping, the process remains focused.

Setting Out Watershed Management Goals

After scoping, the stakeholders have a clearer picture of the issues that they need to address. Therefore, the objectives for the watershed management are set taking into account the issues identified and the resources available. The goals may be broad at the start but will narrow down as the process continues. When setting the goals, it is important to keep in mind that all issues that were identified during the scoping stage may not be addressed at the same time, and therefore it is a good idea to prioritize some issues. An example of a goal at this stage may be to reduce the surface water pollution or developing an equitable water sharing plan in the watershed. The goals may be broad but will help in the next stage of data collection and analysis.

Data Collection and Analysis

As pointed out already, the extent and the quality of data and information required will depend on the objectives and scope of watershed management. The first step in data collection is to gather all the relevant existing data. The sources may include- but not limited to: government departments and institutions, NGOs, universities, research institutions, and credible Internet databases. Data quality assessment is necessary to ensure that the data used for analysis do not lead to wrong conclusions or wrong decisions especially the selection of watershed management strategies. Time series analyses (e.g., trend analysis), for example, require long-term (e.g., over 20 years) observed data series (e.g., of discharge, rainfall, temperature, etc.). This kind of data always requires quality checks (e.g., wrong entries and presence of gaps) before analysis. Details on the causes and how to deal with uncertainties including gap filling of observed climate, discharge, and water quality data are available in literature (e.g., McMillan et al., 2012; WMO, 2012).

Additional data may be collected if it is required and not available. Data collection is also necessary during the monitoring and evaluation stage. The process of data collection should employ the right tools, equipment, and methods to ensure data collected is credible and correct for decision-making. Wrong design and timing of sampling for water quality, for example, could lead to considerable over- and underestimations of, for example, nutrient and sediment loads in the watershed (McMillan et al., 2012; Defew et al., 2013; Jordan and Cassidy, 2011).

With technological advancements, there are several tools and computer programs that are available for data analysis. Geographical information system (GIS) is a powerful tool for the analysis and visualization of spatial data. GIS is particularly useful in distributed rainfall-runoff modelling. It allows input of data with spatial variability (e.g., soils, land use, rainfall) in the models. Watershed delineation and calculation of flow parameters, e.g., flow path length, accumulation, and direction from topographic data (e.g., digital elevation models), are some of the capabilities of GIS applicable in watershed modelling.

Remote sensing is another technology with a wide application in watershed assessment (Ward and Trimble, 2004). Analysis of satellite images and aerial photography is a quick way of getting watershed conditions. One common and important analysis of satellite images is the development of land-use/land cover maps of watersheds through visual or digital image classification techniques (Richards, 2013). This is a quick and economical way of assessing the current status of land use/land cover in the watershed as well as to investigate the land-use/land cover changes over time. Another important application of remote sensing with regard to watershed hydrology is the use of weather satellite to monitor earth-atmosphere systems. Weather satellite data can be analyzed to retrieve information for weather forecast. Meteorological parameters that can be derived from weather satellite data include: precipitation, sea and land surface temperature, radiation, wind, water vapour, clouds, and atmospheric gases, e.g., carbon dioxide. Methods of analysis of these parameters can be found in the literature (e.g., Thies and Bendix, 2011; Li et al., 2013; Trigo et al., 2008; Bellerby, 2004). Information on soil moisture is useful not only in hydrology but also in other applications such as crop production.

Research has shown that there is potential of deriving soil moisture from remote sensing data (e.g., Albergel et al., 2012; Njoku et al., 2003).

Analysis of hydrometeorological data is essential for water resources assessment, conservation, and planning of watershed development. Weather parameters, i.e., precipitation, temperature, evaporation, etc., and streamflow are regularly measured in many watersheds. Analysis of long-term observed climatic and streamflow data shows the water balance in the watershed. Several analyses can be carried on the observed historical data depending on the intended objective. Table 2.2 provides examples of some typical analysis for river discharge and rainfall data. There are more analysis that are applicable for the two (rainfall and discharge) and other climatic variables which can be found in literature (e.g., Maidment, 1993; Ward and Trimble, 2004). The average and variability of the climatic parameters and discharge are equally useful and informative of watershed conditions and can be analyzed using basic descriptive statistical procedures.

Selection of Watershed Management Strategies and Interventions

After data analysis, the existing status of the watershed will be known, and measures to improve or protect the evaluated conditions should be developed. The strategies and related measures should solve the specific problems identified, e.g., for pollution, actions or activities to improve water quality to the required standard should be selected. The strategies may be developed for different spatial scales, e.g., national, watershed, and sub-watershed (farm) level. A portfolio of possible interventions can be developed in a first step and afterwards the most feasible ones selected. Suitable criteria for evaluating the strategies should be developed and agreed upon. Some of the factors to consider are (USEPA, 2008):

- Effectiveness
- Cost
- Acceptance by the stakeholders

One way of evaluating the effectiveness of selected management strategies are numerical computer models (Leavesley, 2005). Such models are powerful tools which can be used for the prediction of desired scenarios where a number of processes are simulated using a number of inputs. However, it is important to note that different models have different capabilities, data requirements, and also limitations. Quality of input data directly affects the model outputs and therefore the results of computer model are as good (or as bad) as the data used. The Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998) is an example of a hydrological model that has been widely used for assessing land management scenarios, also in tropical regions (e.g., Gathenya et al., 2011; Hunink et al., 2012; Garg et al., 2012; Quintero et al., 2009). SWAT can be used to evaluate the effectiveness of best management practices for soil and water conservation on water quality and water yield.

Economic analysis of the strategies is required to determine the cost-effectiveness of the strategies and also assess the cost (of implementation and maintenance/running) and the benefits (short term and long term) of the strategies.

Table 2.2: Examples of typical analysis of streamflow and rainfall data

Typical analysis	**Usefulness**
	Streamflow (discharge) data
Flow duration curves	Water yield assessment; planning and licensing of water diversions from the river, e.g., for water supply, irrigation, hydropower, etc.; reserve (environmental) flow estimation; reservoir sedimentation studies; water quality management, e.g., waste-load allocation
Frequency analysis 1. *Low flow analysis*	Abstraction licensing, waste-load allocations, environmental flow estimation
2. *Flood frequency analysis*	Flood risk assessment, design of hydraulic structures, e.g., dykes, culverts, spillways, etc.
Flood (flow) routing	Flood risk management (e.g., flood warning systems, natural, and man-made waterways transport management
Mass curves	Water storage reservoir design and operation
Hydrographs	Flood risk management (e.g., flood warning systems), design of water control structures and watershed planning
Hydrological modeling and simulation	Prediction of land use and climate change on water resources
Double-mass curves	Checking inconsistency (variation) in observed data record
	Rainfall data
Areal rainfall	Spatial representation of rainfall for planning and hydrological modeling
Intensity-Frequency-Duration (IDF) curves	Peak runoff (flood) estimation, design of flood control structures
Hydrological modeling and simulation	Prediction of land use and climate change on water resources

Development of the Watershed Plan

The watershed management plan is the blueprint for development and management of water resources. It sets out the goals, objectives, and actions for managing the water resources within a specified duration of time (GWP and INBO, 2009). The plan should be actionable with set time frames, specific roles, and responsibilities and the also the financing mechanism.

- The plan should include:
- Watershed description
- Status of the watershed (from water resources assessment)
- Stakeholder analysis
- Watershed management goals and objectives
- Watershed management strategy analysis
- Selected watershed management strategies
- Roles and responsibilities of implementation
- Implementation schedule
- Financing arrangement for implementation including sources of funds
- Monitoring and implementation plan

Implementation of the Watershed Plan

The plan is just a roadmap to watershed management, and therefore it has to be implemented to achieve the set goals. It is wise to have a lead institution or team spearheading the implementation by coordinating the various activities. Capacity development of the individuals', groups', or agencies' implementation activities on the ground is required and should be availed.

Information on the progress of the implementation of the plan should continuously be shared among all the shareholders including those who do not have the implementation responsibility. This makes the process credible and smoother because stakeholders are more likely to support the plan when they perceive it as transparent. Continuous flow of information reduces conflicts and other roadblocks that may appear during the implementation of the plan.

Monitoring and Evaluation

Monitoring and evaluation program is one of the main components of watershed management plans and should be developed during the stage of plan development. The main aim of monitoring and evaluation is to measure the progress of the implementation of the plan towards meeting watershed management objectives and to assess the impact the watershed management efforts are making on the watershed issues identified during the watershed assessment. Monitoring is intended to find out the degree and extent to which the watershed management plan and the selected strategies are changing the state of water resources and the economic, social, and ecological conditions in the watershed (GWP and INBO, 2009).

Monitoring Criteria

When developing the monitoring program during the plan development stage, criteria for measuring progress should be developed as well. It is only against these criteria that the success of watershed management efforts can be assessed. The criteria set ought to be realistic and agreed upon by the stakeholders. Criteria comprise of indicators or targets that should be achieved within specific time frames. The indicators can be qualitative or quantitative depending on the variables or activities to be measured. For example, turbidity can be used as an indicator for sediment load reduction in surface water. Turbidity can be measured by equipment such as turbidity meters which give quantitative values or Secchi disks which is more qualitative. Targets such as the size of total land put under agreed soil and water conservation interventions (including afforestation) within a time frame can also be used to monitor progress. Another indicator may be the annual sedimentation in lakes and reservoirs recorded in dated sediment cores.

Monitoring Programs

Monitoring should be made for the water resources themselves and also for the watershed management efforts. The monitoring programs should take into account these two levels of assessment. Measuring the water resources requires determination of the parameters to be measured and the frequency of measurements. The parameters to be monitored depend with the already-set objectives, and the frequency of measurements should be agreed upon. These measurements come with a cost which should also be included in the budget. The roles and responsibilities of monitoring should also be thought and agreed beforehand. The monitoring team should be credible and diligent in their work to ensure transparency of the process. Monitoring the watershed management efforts is helpful to make sure that the implementation program is on schedule. It also helps to identify and solve any issues that may arise during implementation and if necessary change of tact or modification of some strategies. It is also a learning process where lessons learned are used to improve the planning and implementation process in the subsequent watershed planning and implementation cycles. For this reason, documentation is very crucial because it keeps a record of the process of implementation for reporting and for reference.

Impact Evaluation

Watershed management is done to achieve some objectives, e.g., minimize land degradation or raise efficiency in water use. Therefore, after implementation of the watershed management plans, it is always wise to assess whether the set of strategies developed or the implementation of the plan met the set objectives. It is basically assessing the impact of the implemented strategies on the water resources, livelihood, ecosystem, etc. as set in the objectives. The other aim of evaluation is to get information/lessons that should be used in the improvement of the watershed management program. Evaluation should be carried out on the planning and implementations process and the outcome of the watershed management efforts. Planning and implementation process evaluation should focus on:

- Stakeholder engagement
- Use of resources, e.g., financial and human resources
- Organization and management of the process
- Implementation activities (coordination and their effectiveness)

The evaluation of the outcome of the watershed management should be based on the set objectives and may include impact on:

- Water quality
- Water sharing
- Livelihood improvement, i.e., socioeconomic status

- Environmental protection and rehabilitation, e.g., rehabilitation of degraded lands

Evaluation can be carried out using a variety of methods depending on what is to be assessed, e.g., observations, measurements, focus group discussion, and survey interviews. The results of the monitoring and evaluation process should be well documented and used to make adjustments in the plan.

Revision of the Plan

Watershed management is an iterative and adaptive process where the lessons from one stage or cycle are used to improve the planning and implementation process in subsequent stages and cycles (cf. Figure 2.2). Therefore, the results of monitoring and evaluation should be used to make adjustments in the planning and implementation programs.

2.9 Watershed Assessment: Case Study Sasumua Watershed, Kenya

The Sasumua dam watershed (Figure 2.3) is located in the central highlands of Kenya and supplies 15 % of water used in the capital, Nairobi. The local Water Resources Users Association (Sasumua WRUA) carried out a water abstraction survey with support of Water Resources Management Authority (WRMA). WRMA is the state institution with the legal mandate to oversee water resources management in Kenya. The objectives of the survey were to establish the water resources base in the catchment; establish the level of compliance of water use, allocation, and permit conditions; and document riparian area land-use conditions. The abstraction survey was necessary in this watershed because there were several unregistered abstractions and abstraction exceeded the licensed limit. Therefore this exercise was necessary to estimate naturalized flows for water allocation planning.

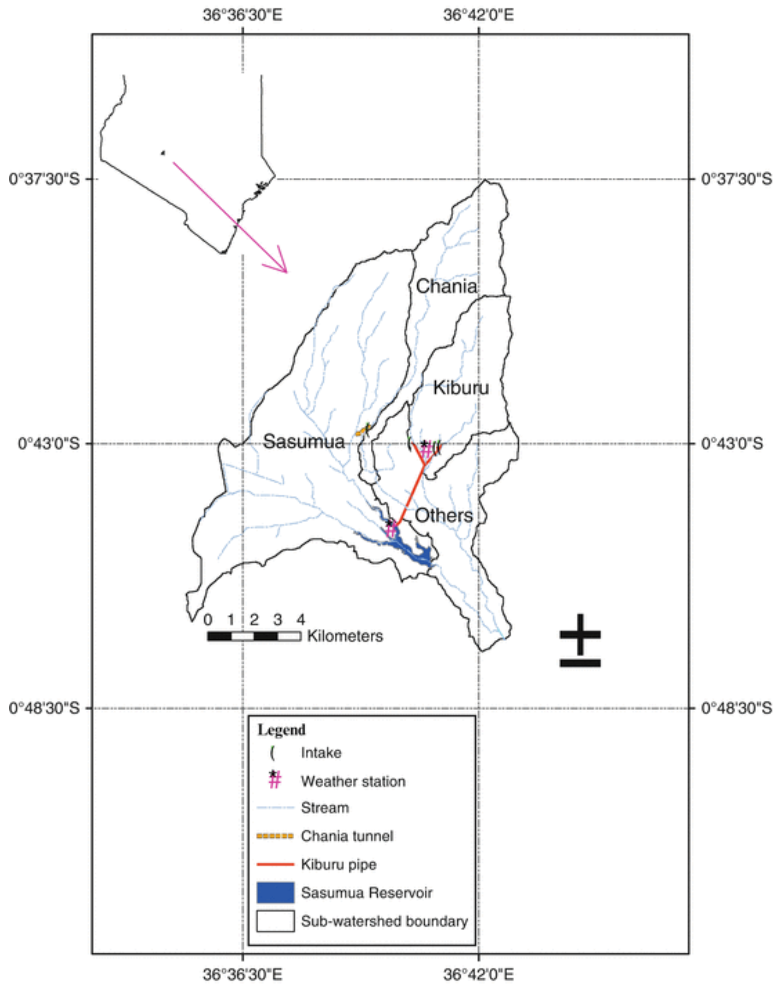

Figure 2.3: Sasumua watershed (Mwangi et al., 2012a)

Prior to the survey, meetings were held to sensitize the local community and especially all the water abstractors on the planned exercise. The meetings, which were organized through the WRUA committee officials and supported by WRMA, were not only important to inform the community of the survey but also to reduce hostility from the community and gain their acceptance and cooperation during the exercise. The meetings also sought to assure the unregistered/illegal abstractors that the exercise was not intended to arrest or prosecute them. Those with abstraction permits were requested to carry them to site during the day of survey exercise of which the schedule was communicated to them early enough.

Existing data such as climatic, discharge, reservoir levels, and water abstraction permits were first collected from various organizations. Elected WRUA officials led the fieldwork which involved measuring of water abstraction from all diversion points which were well known by the local WRUA officials. A combination of methods and equipment was used to measure the water diverted from the river, e.g., current meters, Acoustic Doppler Velocimeter (ADV), bucket and stopwatch, external pipe flow meters, and hydraulic computations from dimensions of flow structures

Engagement of the local communities through their elected officials in the WRUA proved to be very helpful in this exercise. Most of the water abstractors including the unlicensed ones turned up for the exercise and gave the required information. It also gave the opportunity where the WRMA officials met the illegal abstractors (found to comprise 25 % of all abstractors) and informed them of the need to register and apply for water abstraction licenses. The estimated abstractions were found to exceed the permitted amount (Table 2.3). As it was found out, the excess abstraction was not only because of unregistered water users but also contributed by abstraction in excess of permitted limits by licensed abstractors (Mwangi et al., 2012b). Free engagement with the abstractors was very useful in getting this information which otherwise they would not divulge.

Table 2.3: Surface water abstraction status of Sasumua watershed (Mwangi et al., 2012b)

Stream	**Permitted abstraction (m^3/day)**	**Estimated abstraction (m^3/day)**
Chania	1,459	7,599
Kiburu	11,467	12,152
Sasumua	173	173

Collaboration with other institutions working in the watershed was useful to provide data, information, and resources which were necessary for this exercise and also for the entire watershed management. Pro-poor Rewards for Environmental Services in Africa (PRESA) project which is a collaboration between World Agroforestry Centre (ICRAF) and the Jomo Kenyatta University of Agriculture and Technology (JKUAT) provided topographic, climatic, and land-use data. Further, research studies conducted in the watershed under the project had identified soil erosion “hotspots” (Figure 2.4) and identified suitable sustainable land management practices to control degradation of the watershed (Mwangi et al., 2012a, 2014).

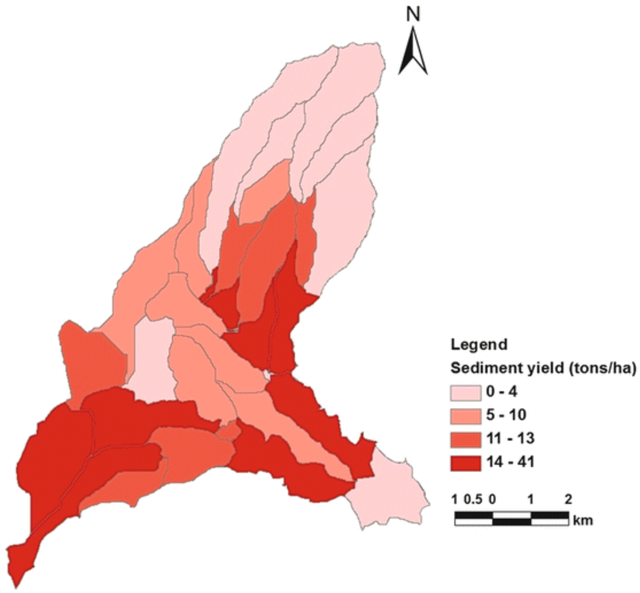

Figure 2.4: Soil erosion “hotspots” in Sasumua watershed based on simulation using the SWAT model (Mwangi et al. 2012a).

3 Chapter three: Land use change intensity of Mara River Basin, East Africa

Publication (this chapter/paper has been submitted to 'Applied Geography' Journal)

Mwangi HM, Lariu P, Julich S, Patil SD, McDonald MA, Feger KH. 2016. Land use change intensity of Mara River Basin, East Africa. Applied Geography (submitted)

Abstract:

The objective of this study was to analyze patterns, dynamics and processes of land use/cover changes in the Mara River Basin from 1976 to 2014. We specifically focused on deforestation and expansion of agriculture in the watershed. The intensity analysis approach was used to analyze data from satellite imagery-derived land use/cover maps. Results indicate that the overall land use/cover change was fastest between 1995 and 2003. Swap change accounted for more than 50% of the overall change at all times, which indicates a very dynamic landscape transformation. Transition from closed forest to open forest was found to be a systematic transition (i.e. a dominant landscape change, as opposed to a random change). Similarly, transition from open forest to small scale agriculture was also found to be a dominant transition. The two transitions were stationary over the entire study period. This suggests a trend (pathway) of deforestation from closed forest to small scale agriculture, with open forest as a transitional land cover. The observed deforestation was attributed to continuous encroachment and a series of excisions of the forest reserve. Transition from rangeland to mechanized agriculture was found to be a dominant land use change between 1985 and 2003, which was attributed to change in land tenure. During the last decade, expansion of mechanized agriculture avoided gaining from rangeland and intensively targeted small scale agriculture, which was attributed to establishment of wildlife associations. These findings are crucial for designing strategies and policies to arrest further deforestation in the forest reserves as well as to sustainably control expansion of agriculture.

3.1 Introduction

Land use/cover change is a topic that has in the recent years gained interest because of its role as a driver of environmental change (Al-doski et al., 2013; Foley et al., 2005). It is of interest to many fields of science including hydrology, biology, environment, biodiversity, conservation and agriculture (Jewitt et al., 2015; Raini, 2009; Alo and Pontius, 2008; Mallinis et al., 2014). The effect of forest-related land use changes e.g. deforestation, afforestation, agroforestry on the flow regime of rivers and water quality has been of special interest to hydrologists and water managers (Mao and Cherkauer, 2009; DeFries and Eshleman, 2004; Gitau and Bailey, 2012; Brown et al., 2013; Mwangi et al., 2016a). Biologists have also been keen to study the effect of land use change on the biodiversity of flora and fauna (de Chazal and Rounsevell, 2009); whereas environmental conservationists are concerned about protecting land uses of high environmental value such as wetlands, forests, green areas in and around cities that have been under increasing threat of conversion to other land uses e.g. agriculture and urban centers (Estoque and Murayama, 2015; Öborn et al., 2015; Huang et al., 2012; Plieninger, 2012).

With increasing world population, the demand for food production continues to rise (Foley et al., 2005). This has led to significant expansion of areas under agriculture, especially in the tropics (Öborn et al., 2015; Lambin and Meyfroidt, 2011). Forests have been the main targets for conversion to agricultural cultivation (Mao and Cherkauer, 2009; Romero-Ruiz et al., 2012; Carmona and Nahuelhual, 2012). Between 2000 and 2005, DeFries et al. (2010) identified urban population growth and agricultural trade as the main drivers of forest loss in the tropics. Gibbs et al. (2010) estimated that 55% of new agricultural land in the tropics between 1980 and 2000 came from intact forests while a further 28% came from disturbed forests. Worldwide, cropland increased from 5.9% to 10.6% of the global land area between 1900 and 2000 (Goldewijk et al., 2011). The demand for housing has also gone up, causing more land to be converted into urban centres and cities to accommodate the needs of the rising population (Mubea et al., 2014; Mundia and Aniya, 2006). Globally, the urban population has risen to over 50% of the world's population from about 10% in the 1900 (Grimm et al., 2008; UN-HABITAT, 2010). Since the same land is still expected to continue providing other ecosystem services (such as water purification, carbon sequestration, air purification and modification of micro-climate, and cultural heritage and leisure (e.g. in parks)), the pressure on land has been increasing steadily and so is the competition from all these land uses (Foley et al., 2005). Therefore, the entire process of land use/cover change, including the patterns, dynamics, and the driving forces, is of interest to a wide range of stakeholders. Indeed, several studies have focused on modelling land use/cover based on the trends of past land use changes and the underlying driving forces in order to predict future land use/cover configurations (Ku, 2016; Mubea et al., 2014; Mas et al., 2014; Verburg et al., 2004). This information is crucial for prediction of possible impacts of the change on human well-being and the environment for timely interventions (Aldwaik and Pontius, 2012; DeFries and Eshleman, 2004).

A good understanding of the land use/cover change processes is fundamental for the establishment of effective conservation and management strategies as well as for modelling future land use (Al-doski et al., 2013; Alo and Pontius, 2008). Superficial assessment of land use/cover change might not reveal the most important and dominant signals of land change, which may lead to wrong choice of conservation measures or inaccurate modelling of land change (Pontius et al., 2004). Overlay of land cover maps (of the same spatial extent), derived from satellite images taken at different time periods, is a commonly used approach of land use/cover change analysis (Mundia and Aniya, 2005). Some studies use transitional matrices to compare land cover maps from two different dates. These kinds of analyses are able to identify the patterns (spatial arrangement of land cover/uses), as well as the magnitude and rates of land use/cover changes. These studies, however, rarely identify or assess the processes behind the change in land patterns i.e. whether the observed transitions are due to systematic or random processes. Identification of these processes aids in linking the observed transitions of land categories to possible causes (Teixeria et al., 2014; Braimoh, 2006; Pontius et al., 2004). In a random process of land change, a given land use category usually has no particular tendency to gain or lose to any of the other categories. If an observed transition deviates from a transition that is expected of a random process, then it is said to be systematic (Alo and Pontius, 2004). Random processes are influenced by coincidental factors, i.e., short-term events of land transformation characterized by rapid and abrupt changes caused by land factors that act suddenly (e.g. land conflicts) (Lambin et al., 2003; Carmona and Nahuelhual, 2012). Systematic processes however tend to evolve in a consistent and progressive way due to factors such as population growth, industrial/commercial expansion or changes in land management policies (Teixeria et al., 2014; Braimoh, 2006; Lambin et al., 2003). Systematic transitions therefore show the most prominent (dominant) signals of landscape change. Thus, characterizing and linking the observed patterns with the processes that cause the transitions help in deeper understanding of land use change and developing effective land management strategies (Mallinis et al., 2014; Huang et al., 2012).

Pontius et al. (2004) developed a method to quantify, detect and differentiate between systematic and random processes of landscape transitions. The method analyzes land use changes relative to the sizes of land use/cover categories. Accounting for size of land use categories helps to identify systematic transitions which might be obscured especially when there are large land use categories which have undergone large changes (Mallinis et al., 2014). This is because large transitions between land categories do not necessarily constitute the most systematic landscape change, as large transitions between the largest land use categories can be expected under a random process of change. (Braimoh, 2006; Pontius et al., 2004). The approach by Pontius et al. (2004), which was recently advanced/improved by Aldwaik and Pontius (2012), is based on land change intensities, i.e., ratio of the size of a change (loss or gain) to the size of the land use/cover category involved in the change. The method first calculates the land use/cover change intensities which would be expected under a random process of change (uniform intensity). Observed change intensities (of gain or loss) are then calculated and compared with the uniform intensity. Simply put, a land cover category is considered to gain (lose) randomly from others if the gains (losses) are in proportion to the availability of the losing (gaining) categories. Large positive or negative deviations from the uniform intensity indicate that systematic transitions (as opposed to random transitions) occurred between two land cover categories (Aldwaik and Pontius, 2012; Braimoh, 2006). The improved approach of Aldwaik and Pontius (2012) provides a unified framework, referred to as intensity analysis, which combines three levels of analysis: the interval, the category and the transition levels. The interval level is the first one and examines how size and speed of change vary across time intervals; the second level (category) examines how the size and intensity of gross losses and gross gains in each category vary across categories for each time interval; the transition level examines how size and intensity of a category's transitions vary across the other categories that are available for that transition. This approach (Intensity analysis) can therefore be used to answer fundamental questions of land use/cover change e.g. when (time interval) was the rate of overall change fastest or slowest? Which land use category is relatively active or dormant and how does it compare across time intervals? Which transitions are intensively targeted or avoided by a particular land use category?

We used the intensity analysis approach of Aldwaik and Pontius (2012) to analyze land use/cover changes in the Mara River basin in East Africa. The land use changes, especially deforestation in the Mau forest at the headwaters, have been blamed for change in the flow regime of the Mara River as well as deterioration of river water quality (Gereta at al., 2009; Kiragu, 2009). Mwangi et al. (2016b) estimated that about 97% of change in the streamflow of Nyangores tributary of the Mara River was caused by land use change, particularly deforestation and expansion of agriculture. A previous land use study by Mati et al. (2008) reported that large scale deforestation occurred in the upper Mara and conversion of rangelands to agriculture in the mid regions of the basin between 1973 and 2000. The Mati et al. (2008) study focused on the net changes of land use and did not investigate the dynamics of land use in details to determine dominant signals of land use change. Analyses based on net changes may fail to reveal the total change on the landscape (Yuan et al., 2016; Fuchs et al., 2015). This is because a zero net change does not necessarily imply a lack of change. There is a possibility that a change occurs in a such a way that the location of a land category changes between time 1 and time 2 while the quantity (size) remains the same (Pontius et al., 2004). For example, analysis of net changes may indicate that a particular land use (e.g. forest) did not change from time point 1 to a subsequent time point 2. However, whereas the size of forest may have remained constant, there could have been deforestation in some parts of the study area which was accompanied by regrowth or afforestation (of equal size) in other parts. This suggests that although the size of forest was constant, the forest was not stable. Revealing this kind of information is important for conservation and management because the changes may have caused a change of hydrological catchment properties (e.g. infiltration properties) which subsequently affects catchment water yield. Our study focused on analysis of transitions among various land use categories at different time intervals. We particularly pay special attention to changes in forest and agriculture with an aim of revealing underlying processes, trends and possible driving forces.

3.2 Material and Methods

3.2.1 Study area

Mara River basin is a transboundary watershed (13,750 km^2) shared between Kenya (65%) and Tanzania (35%) (Figure 3.1). The Mara River originates from the Mau forest complex in Kenya, flows through an expansive rangeland and drains into Lake Victoria on the Tanzanian side of the border. The main land uses are: forest, agriculture and rangeland. Mau is a large forest complex which is a source of several rivers in the region including the Mara River. It is composed of 22 forest blocks which are gazetted forest reserves (NEMA, 2013; GoK, 2009). However, rampant deforestation has occurred in the larger Mau forest complex especially in the last half a century. The area surrounding the Mau forest is favourable for agriculture (annual rainfall ranges from 1400 mm to 2500 mm) and therefore most of the deforested land is converted to agriculture, particularly small scale agriculture. The middle section of the basin is mainly rangeland that is used by the local Maasai people for grazing. Two national (game) reserves are located within the rangeland. The Maasai Mara National Reserve in Kenya and the Serengeti National Reserve in Tanzania are major wildlife tourist attraction sites. In recent years, agriculture, particularly large scale mechanized cultivation of wheat has been extending into the rangelands (Serneels et al., 2001). The downstream section of the basin in Tanzania is mainly dominated by subsistence agriculture and gold mining.

3.2.2 Data

Landsat Multispectral Scanner, Thematic Mapper, Enhanced Mapper Plus and Operational Land imager satellite images for 1976, 1985, 1995, 2003 and 2014 (Table 3.1) were obtained from the United States Geological Survey. The images were chosen for varying dates in the dry season of January and February to avoid clouds and possible errors resulting from seasonal differences between time points.

3.2.3 Land use/cover classification

Image processing was performed using the ArcGIS 10.3.1 software (ESRI, 2015). Supervised classification using maximum likelihood algorithm was adopted using 6 land use/cover categories based on the information from field visits, Google Earth images and available maps of the area (GoK, 1983; FAO, 2002; Mati et al., 2008).

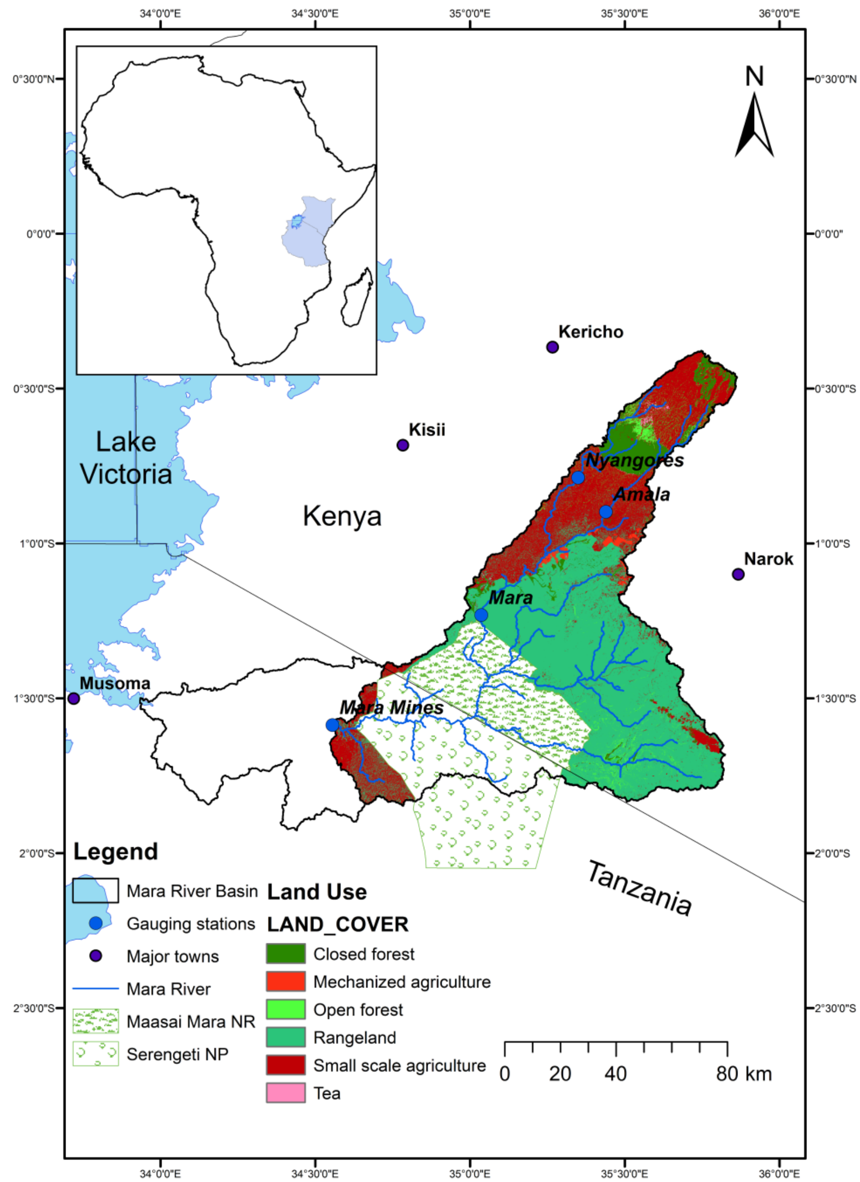

Figure 3.1: Mara River Basin with 2014 land use/cover distribution.

The land cover categories are: closed forest, open forest, small scale agriculture, mechanized agriculture, rangeland and tea plantation. Closed forest represents densely forested areas with closed canopies, while open forest represents areas with light tree canopy coverage and mosaics of predominantly forested areas with some patches of cleared/cultivated land. Small scale agriculture includes patterns of small cultivated areas sometimes alternation with fallow land, while mechanized agriculture comprises of large coherent agricultural land of the same crop stand, mostly in regular shapes. Tea plantation includes both small and large tea plantations. Rangeland includes grassland, shrub land, savannah mainly used for grazing and game reserves. The land use/cover map of 1976 was resampled using nearest neighbour technique from originally 60m to 30m resolution to conform to the resolution of other maps for easier post classification comparison/analysis.

Table 3.1: Satellite Imagery used for LULC classification (P = path, R = Row, FCC = false colour composite).

Nr.	Scene	Sensor	Platform	Pixel size	Bands (FCC)	Date
1	P181 R60	MSS	Landsat 3	60 m	7,5,4	25th Jan 1976
2	P181 R61	MSS	Landsat 3	60 m	7,5,4	12th Feb 1976
3	P182 R61	MSS	Landsat 3	60 m	7,5,4	26th Jan 1976
4	P169 R60	TM	Landsat 5	30 m	4,3,2	9th Jan 1985
5	P169 R61	TM	Landsat 5	30 m	4,3,2	9th Jan 1985
6	P170 R60	TM	Landsat 5	30 m	4,3,2	16th Jan 1985
7	P170 R61	TM	Landsat 5	30 m	4,3,2	16th Jan 1985
8	P169 R60	TM	Landsat 5	30 m	4,3,2	21st Jan 1995
9	P169 R61	TM	Landsat 5	30 m	4,3,2	21st Jan 1995
10	P170 R61	TM	Landsat 5	30 m	4,3,2	12th Jan 1995
11	P169 R60	ETM+	Landsat 7	30 m	4,3,2	4th Feb 2003
12	P169 R61	ETM+	Landsat 7	30 m	4,3,2	4th Feb 2003
13	P170 R61	ETM+	Landsat 7	30 m	4,3,2	10th Jan 2003
14	P169 R60	OLI	Landsat 8	30 m	5,4,3	25th Jan 2014
15	P169 R61	OLI	Landsat 8	30 m	5,4,3	25th Jan 2014
16	P170 R60	OLI	Landsat 8	30 m	5,4,3	16th Jan 2014
17	P170 R61	OLI	Landsat 8	30 m	5,4,3	1st Feb 2014

3.2.4 Land use/cover Intensity analysis

To determine the patterns, intensity and the dynamics of land use /cover change, land use intensity analysis was performed following the framework of Aldwaik and Pontius (2012). First, transitional matrices were used to quantify the land use/cover change over space and time. A transition matrix is a two dimensional table in which land use categories at the beginning of the time interval are displayed in rows while land use categories at end of time interval are displayed in columns (Mallinis et al., 2014, Aldwaik and Pontius, 2012). The diagonal entries indicate persistence whereas off-diagonal entries indicate a transition from one land use category at the start of the interval to another category at the end of the interval. The row totals show the size of land use/cover category at the beginning of the interval and the column totals show the corresponding sizes at the end of the interval. A transitional matrix was made for each time interval i.e. 1976-1985, 1985-1995, 1995-2003 and 2003-2014.

The transition matrices were then used to perform intensity analysis following the approach by Aldwaik and Pontius (2012, 2013). The intensity analysis approach is a top-down approach organized into three levels: the interval level, the category level and the transition level. The method computes and compares observed intensities of land use changes with uniform intensity among intervals and categories. Table 3.2 gives a summary of equations (3.1-3.8) used for intensity analysis at the three levels; the descriptions of the mathematical notations used in the equations are given in Table 3.3.

Interval level

At the interval level, the method analyzes how the size and the annual rate of change vary across time intervals. The interval level analysis compares the observed annual change intensity, $S_{t,}$ (Equation 3.1) to a hypothetical uniform annual rate, U (a rate that would exist if the annual change rates for each interval were distributed uniformly over the entire period of study (Equation 3.2)). The annual change rate is considered fast if $S_t < U$ and slow if $S_t > U$.

Category level

At the category level, the intensity analysis method assess the spatial variation of size and intensity of overall (gross) gains, $G_{tj,}$ and gross losses L_{ti}. For each category, the intensities of gross gains (Equation 3.3) and gross losses (Equation 3.4) are computed and compared with a hypothetical uniform intensity that would exist if the change within each interval were uniformly distributed across the entire study area (which is equal to S_t). Categories that have greater change intensity than S_t are considered active while those whose change intensity is lower than S_t are considered dormant.

Transition level

At the transition level, the method assesses how the size and intensity of the each transition (from one category to the other) varies among land use categories. For each category gain or loss, transition level analysis compares the observed intensity of each transition with a hypothetical uniform transition that would occur if the transition were distributed uniformly among land use categories available for the transition. Equations 3.5 and 3.6 are used for transition level analysis to identify the transition from an arbitrary category *i* to a particular gaining category *n* (Pontius et al., 2013; Aldwaik and Pontius, 2012). In other words, the equations identify which land use categories are intensively avoided or targeted for gaining by category *n* in a particular time interval. As such the analysis can identify which land use categories are intensively targeted (or avoided) by either loss or gain of a particular category. Equation 3.5 calculates observed intensity R_{tin} of annual transition from category *i* to category *n* for a given time interval relative to the size of category *i* at the start of the interval. The observed intensity R_{tin} is compared with uniform intensity W_{tn} calculated using Equation 3.6 which assumes that category *n* gains uniformly across the landscape. If $R_{tin} > W_{tn}$, then gain of *n* is considered to target *i* but if the opposite is true, gain of *n* is considered to avoid *i*. Considering a losing category, and similar to Equations 3.5 and 3.6, Equations 3.7 and 3.8 are used to assess the transition from a particular losing category *m* to a different category *j*. The observed intensity Q_{mtj} is calculated using Equation 3.7 while the hypothetical uniform intensity V_{tm} which would occur if category *m* were to lose at the same annual intensity to all non-*m* categories is calculated using Equation 3.8. If $Q_{mtj} > V_{tm}$ then category *j* is considered to target loss of category *m* whereas if $Q_{mtj} < V_{tm}$ the category *j* is said to avoid loss of category *m*. Therefore, with this analysis, it is possible to tell, for example, which land use categories are targeted (or avoided) by expansion of agriculture. It can also tell if a forest loses, which other land use categories would benefit (targeted) from that loss.

Table 3.2: Summary of equations used in Intensity Analysis

Equation	No.
Interval level equations	
$\boldsymbol{S_t = \frac{change\ during[Y_t, Y_{t+1}]}{(duration\ of\ [Y_t, Y_{t+1}])(domain\ of\ [Y_t, Y_{t+1}])} 100\% = \frac{\sum_{j=1}^{J}[(\sum_{i=1}^{J} C_{tij}) - C_{tij}]}{(Y_{t+1} - Y_t)\left(\sum_{j=1}^{J}\sum_{i=1}^{J} C_{tij}\right)} 100\%}$	3.1
$\boldsymbol{U = \frac{weighted\ sum\ of\ changes\ during\ intervals}{(duration\ of\ [Y_t, Y_{t+1}])(weighted\ sum\ of\ domains)} 100\% = \frac{\sum_{t=1}^{T-1}\left\{(Y_t - Y_{t+1})\sum_{j=1}^{J}[(\sum_{i=1}^{J} C_{tij}) - C_{tjj}]\right\}}{(Y_T - Y_1)\sum_{t=1}^{T-1}\left[(Y_{t+1} - Y_t)(\sum_{j=1}^{J}\sum_{i=1}^{J} C_{tij})\right]} 100\%}$	3.2
Category level equations	
$\boldsymbol{G_{tj} = \frac{size\ of\ annual\ gain\ of\ j\ during\ [Y_t, Y_{t+1}]}{size\ of\ j\ at\ t+1} 100\% = \frac{\left[(\sum_{i=1}^{J} C_{tij}) - C_{tjj}\right]/(Y_{t+1} - Y_t)}{\sum_{i=1}^{J} C_{tij}} 100\%}$	3.3
$\boldsymbol{L_{ti} = \frac{size\ of\ annual\ loss\ if\ i\ during\ [Y_t, Y_{t+1}]}{size\ of\ i\ at\ t} 100\% = \frac{\left[(\sum_{j=1}^{J} C_{tij}) - C_{tii}\right]/(Y_{t+1} - Y_t)}{\sum_{j=1}^{J} C_{tij}} 100\%}$	3.4
Transition level equations	
$\boldsymbol{R_{tin} = \frac{size\ of\ annual\ transition\ from\ i\ to\ n\ during\ [Y_t, Y_{t+1}]}{size\ of\ i\ at\ t} = \frac{C_{tin}/(Y_{t+1} - Y_t)}{\sum_{j=1}^{J} C_{tij}} 100\%}$	3.5
$\boldsymbol{W_{tn} = \frac{size\ of\ annual\ gain\ of\ n\ during\ [Y_t, Y_{t+1}]}{size\ of\ not\ n\ at\ t} = \frac{\left[(\sum_{i=1}^{J} C_{tin}) - C_{tnn}\right]/(Y_{t+1} - Y_t)}{\sum_{j=1}^{J}[(\sum_{i=1}^{J} C_{tij}) - C_{tnj}]} 100\%}$	3.6
$\boldsymbol{Q_{tmj} = \frac{size\ of\ annual\ transition\ from\ m\ to\ j\ during\ [Y_t, Y_{t+1}]}{size\ of\ j\ at\ t} = \frac{C_{tmj}/(Y_{t+1} - Y_t)}{\sum_{i=1}^{J} C_{tij}} 100\%}$	3.7
$\boldsymbol{V_{tm} = \frac{size\ of\ annual\ loss\ of\ m\ during\ [Y_t, Y_{t+1}]}{size\ of\ not\ m\ at\ t+1} = \frac{\left[(\sum_{j=1}^{J} C_{tmj}) - C_{tmm}\right]/(Y_{t+1} - Y_t)}{\sum_{i=1}^{J}[(\sum_{j=1}^{J} C_{tij}) - C_{tim}]} 100\%}$	3.8

Error analysis

The maps used for these analyses were made using satellite images taken at different times in the past. Since the maps cover the same areal extent, if the maps were completely accurate, the differences between them would perfectly indicate temporal changes in land use/cover. However, classification errors in the maps can also cause differences in the maps (Enaruvbe and Pontius, 2015), in addition to the observed land use/cover changes. Since the maps were made from the past images, there was no enough reference ground information available to measure or estimate classification errors in the maps. Aldwaik and Pontius (2013) proposed a methodology for estimating minimum errors in the maps that could account for the differences in two maps of the same areal extent but taken at different time points in the past. The error analysis method by Aldwaik and Pontius (2013) assess the strength of the changes identified through intensity analysis. A null hypothesis of uniform change intensity is first hypothesized at all the three levels of the intensity analysis. Uniform hypothesis assumes that the change intensities are uniform and data errors could account for the deviation between observed intensity and hypothesized uniform intensity.

Table 3.3: Mathematical notation for the intensity analysis equations

Symbol	Meaning
T	Number of time points
Y_t	Year at time point t
t	Index for the initial time point of interval $[Y_t, Y_{t+1}]$, where t ranges from 1 to T-1
J	Number of categories
i	Index for a category at an interval's initial time point
j	Index for a category at an interval's final time point
m	Index for the losing category for the selected transition
n	Index for the gaining category for the selected transition
C_{tij}	Number of elements that transition from category i to category j during interval $[Y_t, Y_{t+1}]$
S_t	Annual change during interval $[Y_t, Y_{t+1}]$
U	Uniform annual change during extent $[Y_t, Y_{t+1}]$
G_{tj}	Intensity of annual gain of category j during interval $[Y_t, Y_{t+1}]$ relative to the size of category j at time t+1
L_{ti}	Intensity of annual gain of category i during interval $[Y_t, Y_{t+1}]$ relative to the size of category i at time t
R_{tin}	Intensity of annual transition from category i to category n during interval $[Y_t, Y_{t+1}]$ relative to the size of category i at time t
W_{tn}	Uniform intensity of annual transition from all non-n categories to category n during interval $[Y_t, Y_{t+1}]$ relative to the size of all non-m categories at time t
Q_{tmj}	Intensity of annual transition from category m to category j during interval $[Y_t, Y_{t+1}]$ relative to the size of category j at time t+1
V_{tm}	Uniform intensity of annual transition from category m to all non-m categories during interval $[Y_t, Y_{t+1}]$ relative to the size of all non-m categories at time t+1
$E^S{}_t$	Commission of change error during interval $[Y_t, Y_{t+1}]$, i.e., percent of domain that is observed change during interval $[Y_t, Y_{t+1}]$ but is hypothesized persistence
$O^S{}_t$	Omission of change error during interval $[Y_t, Y_{t+1}]$, i.e., percent of domain that is observed persistence during interval $[Y_t, Y_{t+1}]$ but is hypothesized change
$E^G{}_{tj}$	Commission of category j error at time t+1, i.e., number of elements that are observed gains of category j during interval $[Y_t, Y_{t+1}]$ but are hypothesized gains of a non-j category
$O^G{}_{tj}$	Omission of of category j error at time t+1, i.e., number of elements that are observed gains of a non-j category during interval $[Y_t, Y_{t+1}]$ but are hypothesized gains of category j
$E^L{}_{ti}$	Commission of category i error at time t, i.e., number of elements that are observed losses of category i during interval $[Y_t, Y_{t+1}]$ but are hypothesized losses of a non-i category
$O^L{}_{ti}$	Omission of of category i error at time t, i.e., number of elements that are observed losses of a non-i category during interval $[Y_t, Y_{t+1}]$ but are hypothesized losses of category i
$E^R{}_{tin}$	Commission of category i error at time t, i.e., number of elements that are observed transtions from category i to category n during interval $[Y_t, Y_{t+1}]$ but are hypothesized transitions from a non-i category to category n
$O^R{}_{tin}$	Omission of category i error at time t, i.e., number of elements that are observed transtions from a non-i category to category n during interval $[Y_t, Y_{t+1}]$ but are hypothesized transitions from a category i to category n
$E^Q{}_{tmj}$	Commission of category j error at time t+1 i.e., number of elements that are observed transitions from category m to category j during interval $[Y_t, Y_{t+1}]$ but are hypothesized transitions from category m to a non-j category
$O^Q{}_{tmj}$	Omission of category j error at time t+1 i.e., number of elements that are observed transitions from category m to a non-j category during interval $[Y_t, Y_{t+1}]$ but are hypothesized transitions from category m to category j

The minimum hypothetical errors that could account for the deviations between the observed and hypothesized change intensity are then calculated. Ideally, if data were perfectly correct, then error would not account for the deviation. In this analysis, two types of errors (commission errors and omission errors) are estimated. Commission error arises when the observed intensity of change is greater than the uniform hypothesized intensity. The reverse is true for the omission error. The error (either commission or omission) is the difference between the observed change and uniform change. The larger the hypothetical commission or omission error, the stronger the evidence against the null hypothesis of uniform change. The strong evidence implies that there is a high likelihood that the differences in the maps are not necessarily due to classification errors; implying that there was actual change in land use. The intensity of commission of error is a compliment of User's accuracy (100% minus User's accuracy) while the intensity of omission error is a compliment of Producer's accuracy (100% minus Producer's accuracy) (Enaruvbe and Pontius, 2015; Pontius et al., 2013).

Table 3.4 gives the summary of the equations (3.9-3.28) used for error analysis at all the three levels. For intervals where $S_t > U$, Equation 3.9 is used to calculate the commission error as a percent of the study area while Equation 3.10 is used to calculate the commission of change intensity during a particular interval. Similarly for intervals where $S_t < U$, Equations 3.11 and 3.12 calculate the omission error and the omission of change intensity respectively. At the category level, Equations 3.13 and 3.14 are used to calculate the size and intensity of commission of category *j* error respectively for active categories whose observed gain of category *j* is greater than uniform change during the interval. Similarly, Equations 3.15 and 3.16 are used to calculate the size and intensity of omission of category *j* error respectively for dormant categories whose observed gain of category *j* less than uniform change during the interval. Similar logic is used to calculate the size and intensity of commission (and omission) of category *i* error in case of observed loss of category *i*. Equation 3.17 and 3.18 are equivalent of equations 3.13 and 3.14 while Equations 3.19 and 3.20 are equivalent of Equations 3.15 and 3.16; the difference being that Equations 3.17, 3.18, 3.19 and 3.20 focuses on observed loss (instead of observed gain) of category *i*.

Table 3.4: Summary of equations for error analysis

Equations for error analysis	
Interval level	
Commission error (% **of study area**) $= \quad \boldsymbol{E_t^S = (S_t - U)(Y_{t+1} - Y_t)}$	3.9
commission of change intensity during $[Y_t, Y_{t+1}] = \frac{E_t^S}{S_t(Y_{t+1}-Y_t)}\, 100\%$	3.10
Omission error (% **of study area**) $= \quad \boldsymbol{O_t^S = (U - S_t)(Y_{t+1} - Y_t)}$	3.11
omission of change intensity during $[Y_t, Y_{t+1}] = \frac{O_t^S}{S_t(Y_{t+1}-Y_t)+O_t^S}\, 100\%$	3.12
Category level	
Gain category	
commission of category j error at $t+1 = \quad E_{tj}^G = \frac{\left(\sum_{i=1}^{J} C_{tij}\right)(Y_{t+1}-Y_t)(G_{tj}-S_t)}{100\% - (Y_{t+1}-Y_t)S_t}$	3.13
commission of j intensity at $t+1 = \frac{E_{tj}^G}{\left(\sum_{i=1}^{J} C_{tij}\right)-C_{tjj}}\, 100\%$	3.14
omission of category j error at $t+1 = \quad O_{tj}^G = \frac{\left(\sum_{i=1}^{J} C_{tij}\right)(Y_{t+1}-Y_t)(S_t-G_{tj})}{100\% - (Y_{t+1}-Y_t)S_t}$	3.15
omission of j intensity at $t+1 = \frac{O_{tj}^G}{\left[\left(\sum_{i=1}^{J} C_{tij}\right)-C_{tjj}\right]+O_{tj}^G}\, 100\%$	3.16
Loss category	
commission of category i **error at** $t \quad E_{ti}^L = \frac{\left(\sum_{j=1}^{J} C_{tij}\right)(Y_{t+1}-Y_t)(L_{ti}-S_t)}{100\% - (Y_{t+1}-Y_t)S_t}$	3.17
commission of i **intensity at** $t = \frac{E_{ti}^L}{\left(\sum_{j=1}^{J} C_{tij}\right)-C_{tjj}}\, 100\%$	3.18
omission of category i **error at** $t = \quad O_{ti}^L = \frac{\left(\sum_{j=1}^{J} C_{tij}\right)(Y_{t+1}-Y_t)(S_t-L_{ti})}{100\% - (Y_{t+1}-Y_t)S_t}$	3.19
omission of i **intensity at** $t = \frac{O_{ti}^L}{\left[\left(\sum_{j=1}^{J} C_{tij}\right)-C_{tii}\right]+O_{ti}^L}\, 100\%$	3.20
Transition level	
Transition from i given observed gain of n	
commission of category i **error at** $t = \quad E_{tin}^R = \frac{\left(\sum_{i=1}^{J} C_{tij}\right)(Y_{t+1}-Y_t)(R_{tin}-W_{tn})}{100\% - (Y_{t+1}-Y_t)W_{tn}}$	3.21
commission of i **intensity at** $t = \frac{E_{tin}^R}{C_{tin}}\, 100\%$	3.22
Omission of category i **at** $t \quad = \quad O_{tin}^R = \frac{\left(\sum_{j=1}^{J} C_{tij}\right)(Y_{t+1}-Y_t)(W_{tn}-R_{tin})}{100\% - (Y_{t+1}-Y_t)W_{tn}}\, 100\%$	3.23
omission of i **intensity at** $t = \frac{O_{tin}^R}{O_{tin}^R + C_{tin}}\, 100\%$	3.24
Transition to j, given observed loss of m	
Commission of category j **error at** $t+1 = \quad E_{tmj}^Q = \frac{\left(\sum_{i=1}^{J} C_{tij}\right)(Y_{t+1}-Y_t)(Q_{tmj}-V_{tm})}{100\% - (Y_{t+1}-Y_t)V_{tm}}$	3.25
commission of i **intensity at** $t+1 = \frac{E_{tmj}^Q}{C_{tmj}}\, 100\%$	3.26
Omission of category j **error at** $t+1 = \quad O_{tmj}^Q = \frac{\left(\sum_{i=1}^{J} C_{tij}\right)(Y_{t+1}-Y_t)(V_{tm}-Q_{tmj})}{100\% - (Y_{t+1}-Y_t)V_{tm}}\, 100\%$	3.27
omission of j **intensity at** $t+1 = \frac{Q_{tmj}}{Q_{tmj}+C_{tmj}}\, 100\%$	3.28

Error analysis at the transition level is carried out using Equations 3.21 to 3.28. The analysis is carried out in two parts. The first one being transition from category *i* given observed gain of category *n*. The second part is transition to category *j* given observed loss of category *m*. For the first part, when observed transition of *i* is greater than uniform gain of *n*, the size and the intensity of commission of category *i* error are estimated using Equations 3.21 and 3.22 respectively. Similarly, the size and the intensity of omission of category *i* error are calculated using Equations 3.23 and 3.24 respectively when observed transition from *i* is less than uniform gain of *n*.

For the second part (transition to category *j* given observed loss of category *m*), when transition to *j* is greater than uniform loss of category *m*, the size and the intensity of commission of category *j* error are estimated using Equations 3.25 and 3.26 respectively. Equations 3.27 and 3.28 are used to calculate the size and intensity of omission of category *j* error when observed transition to *j* is less than uniform loss of *m*.

3.3 Results and Discussion

3.3.1 Overall Land use/cover changes

Figure 3.2 shows the land use/cover maps of the Mara River basin (outlet taken at Mara mines gauging station; Figure 3.1) for the years 1976, 1985, 1995, 2003 and 2014 respectively. From the maps, it can be seen that rangeland is a dominant land use accounting to over two-thirds of the basin at all times. This is partly contributed by the presence of two protected national reserves, i.e., Maasai Mara and Serengeti, whose land cover fall under this land use category. The two national reserves have been in existence throughout the study period. The area surrounding the national reserves, particularly of the Kenyan side of Maasai Mara is occupied by Maasai ethnic community whose main socio-economic activity is pastoralism. Therefore, the land surrounding the Maasai Mara is mainly used for grazing. It can be seen that most of the land use change have occurred in the upper part of the basin. The most notable change is the steady increase in small scale agriculture and a decline in forest cover (Figure 3.2). The forest cover reduced from about 20% to about 7.5 % of the study area between 1976 and 2014, which can be attributed to deforestation particularly in the Mau forest at the source of the Mara River. During the same period, small agriculture increased from approximately 8.5% to 21% of the landscape. Other land use changes of interest are expansion of mechanized agriculture and tea. Although they both represent a very small percentage of the basin (<1%) at all times, they have been steadily increasing during the study period. Mechanized farming was only almost insignificant in

1976 (covered ca. 0.01% of the watershed) while tea is a new land use established in the 1985-1995 interval.

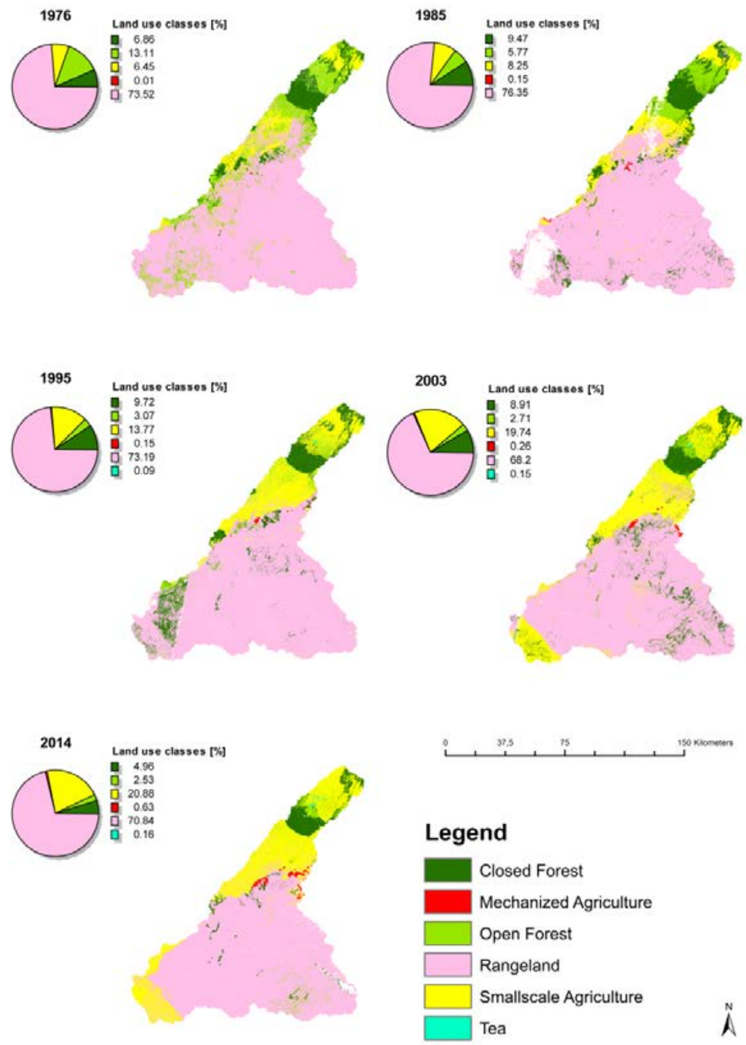

Figure 3.2: Land use/cover maps.

Overall and category-wise annual areas of gross gain, gross loss, net changes and swap changes are summarized in Table 3.5 for the four time intervals. Closed forest shows a net gain (23.3 km^2/year) during the first interval and then shows a net loss in all the other subsequent intervals. Open forest depicts a net loss over the entire study period (during all the intervals). This indicates a general trend of deforestation throughout the study period. Small scale agriculture shows a net gain in all intervals which peaked (73.9 km^2/year) during the 1995-2003 interval and slowed (6.6 km^2/year) during the last interval. The overall land use/cover change also peaked during the 1995-2003 interval (at 258 km^2/year) and slowed during the last interval. Most land use categories underwent swap change throughout the study period (Table 3.5). A swap change depicts vegetation (or any other land use/cover) loss occurring in one location while an equal gain occurs in another location (Yuan et al., 2016; Zaehringer et al., 2015; Pontius et al., 2004).

Table 3.5: Categories of land change (annual change in km^2) during the four intervals

	Gross gain	**Gross Loss**	**Total Change**	**Net Change**	**Swap Change**	**swap change (% of total change)**
			1976-1985			
Closed forest	42.45	19.16	61.61	23.29	38.32	62.20
Open forest	22.30	102.89	125.19	80.60*	44.60	35.62
Small scale agriculture	54.10	34.05	88.15	20.05	68.11	77.26
mechanized agriculture	1.68	0.13	1.81	1.55	0.26	14.44
Rangeland	80.42	44.71	125.13	35.71	89.43	71.46
Tea	0.00	0.00	0.00	0.00	0.00	
Overall Change	200.95	200.95	200.95	80.60	120.36	59.89
			1985-1995			
Closed forest	28.71	39.10	67.81	10.39*	57.42	84.68
Open forest	18.21	43.85	62.06	25.64*	36.42	58.68
Small scale agriculture	80.71	24.35	105.06	56.36	48.70	46.35
mechanized agriculture	1.42	1.33	2.75	0.08	2.67	96.96
Rangeland	40.46	61.53	102.00	21.07*	80.93	79.34
Tea	0.66	0.00	0.66	0.66	0.00	0.00
Overall Change	170.17	170.17	170.17	57.10	113.07	66.44
			1995-2003			
Closed forest	48.73	59.56	108.28	10.83*	97.45	90.00
Open forest	30.72	35.91	66.63	5.20*	61.44	92.21
Small scale agriculture	111.11	37.21	148.32	73.90	74.42	50.18
mechanized agriculture	2.53	1.03	3.56	1.49	2.07	58.05
Rangeland	63.77	123.94	187.71	60.17*	127.54	67.94
Tea	1.67	0.86	2.53	0.80	1.73	68.21
Overall Change	258.51	258.51	258.51	76.20	182.32	70.53
			2003-2014			
Closed forest	7.21	42.01	49.22	34.80*	14.43	29.31
Open forest	16.32	17.89	34.22	1.57*	32.65	95.42
Small scale agriculture	49.67	43.04	92.72	6.63	86.08	92.85
mechanized agriculture	4.24	0.96	5.20	3.28	1.92	36.94
Rangeland	62.34	35.90	98.24	26.44	71.80	73.09
Tea	1.09	1.07	2.15	0.02	2.14	99.25
Overall Change	140.88	140.88	140.88	36.37	104.51	74.19

The last column (Table 3.5) shows that swap change accounted for more than 50% of the overall land use/cover change in all the intervals. Individual land use/cover categories also show that swap change accounted for more than half of the total change in most land use/cover categories (Table 3.5). This shows that land use change in the Mara is very dynamic which is accompanied by high rate of relocation of land use categories. The high percentage of swap change indicates that overall change is more than double the net changes, and shows the importance of detailed analysis of land change beyond the net change (Yuan et al., 2016). For deforestation as an example, analysis of net changes alone would indicate a loss rate of 10.83 km^2/year for closed forest for the 1995-2003 interval. This however may not give the entire picture because this only represents 10% of the total change of closed forest during this interval. The closed forest actually lost at a higher rate of about 60 km^2/year but it also gained (probably by regrowth in formerly opened up areas) by 49 km^2/year. This shows that 90% of the overall change (swap change) would have been concealed and the actual rate of deforestation (particularly of indigenous forest) may be underestimated.

The interval level analysis results (Figure 3.3) shows that the annual rate of land use change was fastest during the 1993-2003 period where there was change of land use in 21% of the study area. The annual change intensity (2.6% of the landscape) for this eight-year period is larger than the uniform intensity (1.91% of the landscape). For this interval, the commission of change error in 6% of the study area with an intensity of commission of change error of 27% of the observed change during the 1995-2003 interval gives strong evidence that the annual rate of change was indeed faster than uniform change (Figure 3.3); implying that the change was not necessarily solely due to map errors. Land use change was slowest during the last interval (2003-2014). The rates of loss of closed forest, loss of rangeland, gain of closed forest and gain of small scale agriculture were also highest (peak) during the 1995-2003 interval (Table 3.5). From their trends, it implies that deforestation, loss of rangelands and expansion of agriculture are the main cause of relatively fast land use change during the 1995-2003 interval.

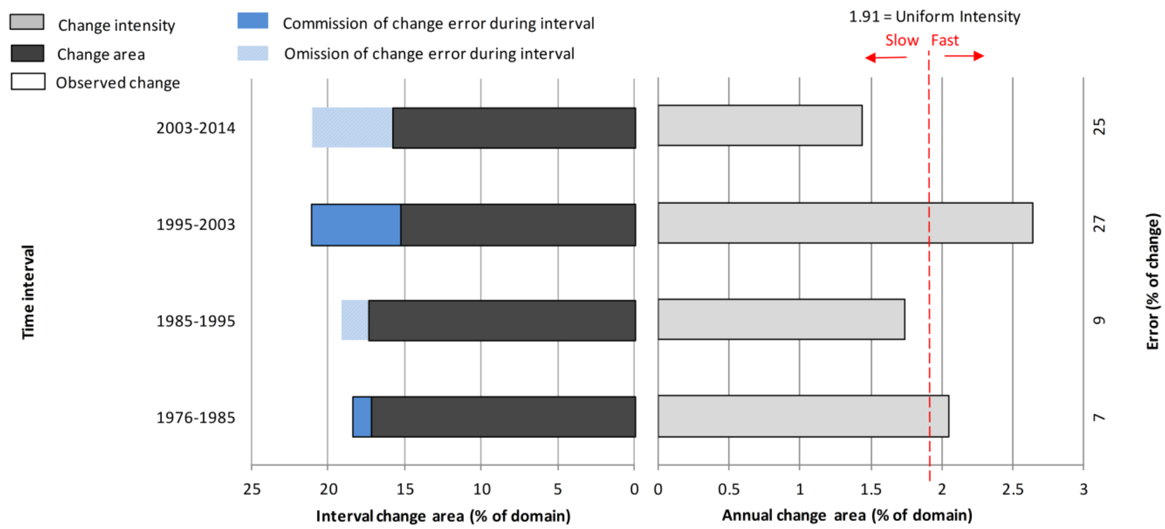

Figure 3.3: Intensity analysis for interval-level change.

The category level analysis also shows that loss of forest (closed and open) was active between 1995 and 2003 i.e. the intensity of forest loss was higher than the uniform intensity (2.64%) (Table 3.6); the intensities of forest loss (as a percent of respective land category in 1995) were 6.1% and 11.7 % for closed forest and open forest respectively. This means that forest experienced change more intensively than if the overall change were to be distributed uniformly across the landscape. The high percentage values in the error columns of Table 3.6 provide stronger evidence against the uniform hypothesis. This pattern (of active forest loss) was stationary across all the intervals (Table 3.6) which imply that forest was actively loosing throughout the study period (1976-2014). The values presented in Table 3.6 could be presented in graphical form as in Figure 3.4 which clearly shows forest (closed and open), and agriculture (mechanized and small scale) as active categories while rangeland is a dormant category when considering category losses during 1985-1995 period. Rangeland was a dormant category during the entire study period (Table 3.6; Figure 3.4) which can be attributed to its large size relative to the other land use categories. Although it loses at a higher annual rate than the other land use categories (e.g. Figure 3.4), its large size lowers the fraction of its size that is lost therefore making it dormant when compared to the fractions of other land use categories that underwent change. In other words, rangeland lost less intensively than the other land use categories.

Presence of a large dormant category generally causes the intensities of other categories to be greater than they would be in its absence mainly because of its possible large persistence (Pontius et al., 2013). Change in rangeland, however, plays a major role in the total land change in the Mara River Basin and cannot be ignored or excluded. For example, during the 1995-2003 interval, the gross loss of rangeland consisted 48% of overall change and its (rangeland) gross gain constituted 25% of the overall change (Table 3.5). In terms of the total size of the landscape, loss and gain of rangeland occurred in about 10% and 5% of the study area respectively, compared with the overall change which occurred in 21% of the study area (Figure 3.3). Furthermore, one of the specific objectives of this study was to assess the extent and nature of expansion of mechanized agriculture in the rangeland (discussed under transition level results) and therefore excluding the rangeland would miss these important transitions. The gain in small scale agriculture was active (more intensive than uniform change e.g. Figure 3.4) and stationary (active in all the intervals) (Table 3.6) which shows small scale agriculture has been continually intensively gaining throughout the study period. The results from transition level analysis give more information on transition between land categories. We particularly focus on transitions among forest, agriculture and rangeland to get more details on deforestation and expansion of agriculture in the study area.

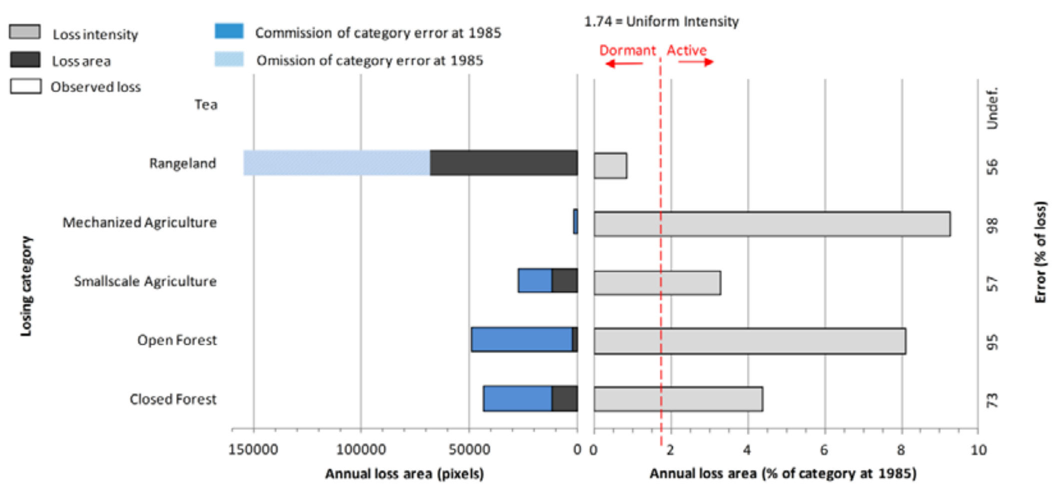

Figure 3.4: Gross loss by categories for the 1985-1995 interval.

Table 3.6: Intensities of gains and losses

	1976-1985		1985-1995		1995-2003		2003-2014	
a) Category gains								
	UI= 2.05		UI= 1.74		UI = 2.64		UI = 1.44	
	GI	EI	GI	EI	GI	EI	GI	EI
Closed forest	4.63	68.3	3.66	63.59	5.49	65.85	**1.4**	(3.12)
Open forest	4.1	61.34	6.4	88.18	11.59	97.89	6.2	91.26
Small scale agric	6.69	85.03	6.19	87.07	5.67	67.79	2.35	46.04
Mechanized agric	11.11	100	9.3	98.41	9.41	91.22	6.81	93.7
Rangeland	**1.07**	(52.91)	**0.58**	(70.94)	**0.94**	(69.72)	**0.86**	(44.6)
Tea			10	100	10.87	95.99	6.35	92.62
b) Category losses								
	UI=2.05		UI= 1.74		UI = 2.64		UI = 1.44	
	LI	EI	LI	EI	LI	EI	LI	EI
Closed forest	2.71	29.77	4.4	73.26	6.11	72.06	4.68	82.28
Open forest	8.11	91.62	8.11	95.09	11.71	98.2	6.38	92.02
Small scale agric	5.42	76.22	3.29	57.14	2.72	3.81	2.11	37.72
Mechanized agric	11.11	100	9.26	98.31	6.94	78.54	3.67	72.2
Rangeland	**0.62**	(73.82)	**0.85**	(55.81)	**1.7**	(41.16)	**0.51**	(68.1)
Tea					9.69	92.25	6.5	92.51

Values in bold indicate dormant land categories where observed intensity is larger than uniform intensity (mostly rangelands). UI is uniform intensity (% of size of corresponding category at the beginning of the interval), GI is gain intensity (% of size of corresponding category at the end of interval) and LI is loss intensity (% of size of corresponding category at the beginning of interval)

3.3.2 Deforestation in the study area

Loss of closed forest was found to have peaked during the 1995-2003 period (Table 3.6). The results from transition level analysis show that closed forest lost more intensively to the open forest than any other land use category during the 1995-2003 interval (Figure 3.5; Table 3.7a). The annual intensity of loss from closed forest to open forest is 3.5% of size of closed forest at 2003, compared to uniform intensity of 0.67% (of the landscape that was not closed forest at 2003). In other words, when closed forest lost, it lost to open forest at a rate over 5 times more than the rate it would be expected to lose randomly (i.e. uniformly to all other land use categories). This is a systematic transition which indicates that when closed forest lost, it tended to lose more intensively to open forest. There is a possibility that a land use category loses systematically to other categories but experiences a random gain from these categories at the same time, simply because factors that promote gain of land use/cover change may be quite different from those that lead to losses (Teixeria et al., 2014; Braimoh, 2006). Therefore, to arrive at a conclusive evidence of a dominant signal of landscape transformation, a given land use category (X) must systematically lose to a second land use category (Y), and the second land use category (Y) must systematically gain from the first land use category (X) simultaneously (Braimoh 2006; Alo and Pontius, 2004, 2008). For our case, the transition to open forest also indicates that open forest intensively targeted closed forest for takeover during this interval (Figure 3.6; Table 3.7c). Therefore, closed forest lost systematically to open forest while at the same time open forest gained systematically from closed forest, which shows a true and strong systematic transition. Systematic transitions show dominant signals of landscape transformation and therefore this transition from closed forest to open forest is a prominent process of land use change (deforestation), as opposed to a random process. The observed systematic process of closed forest transition was stable across all the other intervals (Table 3.7a and 3.7c). This suggests that the closed forest has been consistently and systematically losing to open forest throughout the study period. The error intensity (EI) columns (Table 3.7) present a strong evidence that, indeed, closed forest lost more intensively to open forest i.e. the observed deviations from uniform intensity is not largely due to map errors. This is because if map errors could account for deviations between observed intensities and the uniform intensity, then the evidence for deviations from uniform intensity is weak (Enaruvbe and Pontius, 2015). On the other hand, the evidence for deviations from uniform intensity is strong if the errors in the data cannot account for deviations between observed intensities and corresponding uniform intensity. In our results, for example, the commission error intensity for transition from closed forest to open forest during the 1995-2003 interval is 85.4% of transition from closed to open forest; this high percentage of error intensity indicates a strong evidence against the null hypothesis that closed forest lost uniformly to all the other land use categories. A possible question would be: how large should the deviation (from uniform) be to qualify as real (real change)? First, the method by Aldwaik and Pontius (2013) calculates how much hypothetical error in maps could account for deviations. If data were perfectly correct, then error would not account for deviations (because there would be no map errors). Therefore, if actual error is smaller than the corresponding hypothetical error, then the deviation qualifies as real (Enaruvbe and Pontius, 2015; Pontius et al., 2013). Since actual error is not precisely known, the method does not provide a threshold of deviation for real change. It however implies that the more the hypothetical error, the more the likelihood that the deviation is real. Results of transition from open forest (Table 3.7b), shows that loss of open forest targeted small scale agriculture (open forest lost to small scale agriculture) across all the intervals. Transition to small scale agriculture also targeted open forest (small scale agriculture gained from open forest) across all the intervals also (Table 3.7d). This also indicates a stationary (similar across all intervals) systematic transition from open forest to small scale agriculture.

These systematic processes of deforestation can be attributed to encroachment, excisions and illegal extraction of wood products from the forests especially the Mau forest at the headwaters of the Mara River Basin (GoK, 2009). There has been continuous and progressive encroachment into the Mau forest leading to massive degradation. In the larger Mau forest, for example, over a period of 15 years (1993-2009), a total of 28,500 ha (19 Km^2/year) of forest was lost through encroachment (GoK, 2009). There have also been a series of excisions (degazettement) of forest reserves in the watershed (e.g. Mau forest and Chepalungu forest) since Kenya's independence in 1963 (GoK, 2009; Matiru 1999). The excisions of the forest have also been accelerating encroachment in the remaining forest reserves (GoK, 2009). There was a large scale excision of Mau forest in 2001 which caused much public outcry (Nkako et al., 2005; Akotsi and Gachaja, 2004; GoK, 2009; NEMA, 2013). The excisions affected many parts of the larger Mau forest complex including the Eastern and South West Mau forest blocks which are partly located in the MRB (NEMA, 2013). An aerial survey conducted jointly by United Nations Environment Programme (UNEP) and Government of Kenya (UNEP and GoK, 2008) found that about 35, 301 ha (353 km^2) and 22, 797 ha (228km^2) were excised in Eastern Mau and South Western Mau forest blocks respectively in 2001. This excision of 2001 coincides with the period of fastest land use change observed in our study (i.e. 1995-2003 interval) and may therefore have partly contributed to the fast change in land use observed between 1995 and 2003. The 2001 Mau forest excision could also explain the high loss of closed forest observed in the 1995-2003 interval compared to other intervals (Table 3.7). The Maasai Mau forest block also underwent intensive deforestation during the same period particularly the western side that is located in the Mara River Basin (Nkako et al., 2005). A study by Nkako et al. (2005) found that it (Maasai Mau forest block) lost over 6,300 ha between 2000 and 2005 inside the forest reserve.

Political interference, weak law enforcement, limited management capacities of mandated institutions, and inadequate governance systems are some of the factors that may have led to the large scale and consistent deforestation through encroachment and forest excision (GoK, 2009; Were et al., 2013). A study by Petursson et al. (2013) in Kenya and Uganda found that complex political and institutional factors as the main driving forces behind deforestation. Some of the excisions in the Mau forest were politically ill-motivated and poorly planned leading to irregular allocations (GoK, 2009).

The results (Table 3.7) show a consistent trend of deforestation from closed forest to open forest and then from open forest to small scale agriculture, with open forest being a transition land cover in a deforestation process from closed forest to agriculture. This pathway of deforestation is very important for conservation because the deforestation observed in the Mau forest is mainly carried out illegally by encroaching into the forest reserve. It indicates that the closed forests are first opened (possibly for timber and charcoal) and then the opened patches are cultivated; eventually, the remaining trees are cleared and converted to agricultural land. As a transition land cover, the results show open forest as a very dynamic land cover particularly in the last two intervals (Table 3.7). The ratio (percentage) of swap change to the overall change (Table 3.5) has been increasing steadily over the entire study period; the swap change was about 95% of the total change in open forest during the last interval (2003-2014). This implies that as open forest was gaining from closed forest through encroachments/excisions, it was equally losing to other land use categories (e.g. small scale agriculture and tea) in other parts of the watershed that were opened earlier on. This shows the importance of open forest as a transitional land use in the deforestation process. Our findings/observations are supported by Olang and Kundu (2011) who observed widespread charcoal burning in the Mau forest followed by conversion of deforested areas into subsistence agriculture. An aerial survey by Nkako et al. (2005) also found widespread charcoal burning in freshly opened areas of the Maasai Mau forest block especially on the western side of the forest block which is located in the MRB. Were et al. (2013) also noted that illegal logging and charcoal burning of indigenous forests precedes extensive cultivation in former forest land in the Eastern block of the Mau forest. There is a possibility that the people opening up the forest for timber or charcoal are different from the ones who later cultivate and settle in the former forest land. The people opening up the forest may be timber and charcoal traders operating far from the forest but who have the resources, connections (with authorities and local people), and networks to facilitate harvesting (including deep in the thick forest), transport and sale of these products (Standing and Gachanja, 2014; Cavanagh et al., 2015). These traders may have no intention or interest in the forest land, but their actions lead to opening-up of the forest thus giving the local community easier access. This makes it easier for the local community to start cultivating on the cleared patches of the forest and eventually clearing the remaining trees as they expand 'their' farms. Nkako et al. (2005), for example, observed that many destructive activities in the Mau forest were in close proximity of the forest tracks. This implies that one possible way of combating deforestation is to stop illegal logging for timber and charcoal. In cases where some patches of the closed forest have already been opened-up, measures should be put in place to prevent cultivation of these areas by the local communities.

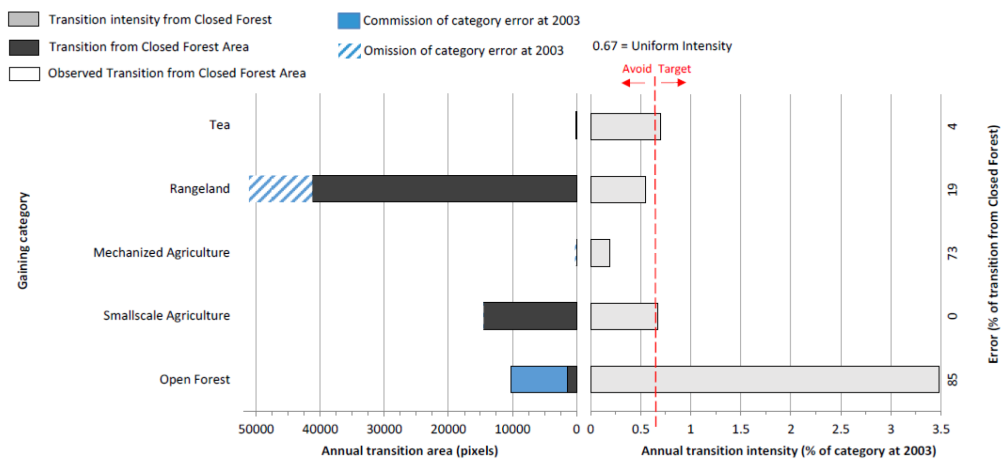

Figure 3.5: Transition from the closed forest (1995-2003 interval).

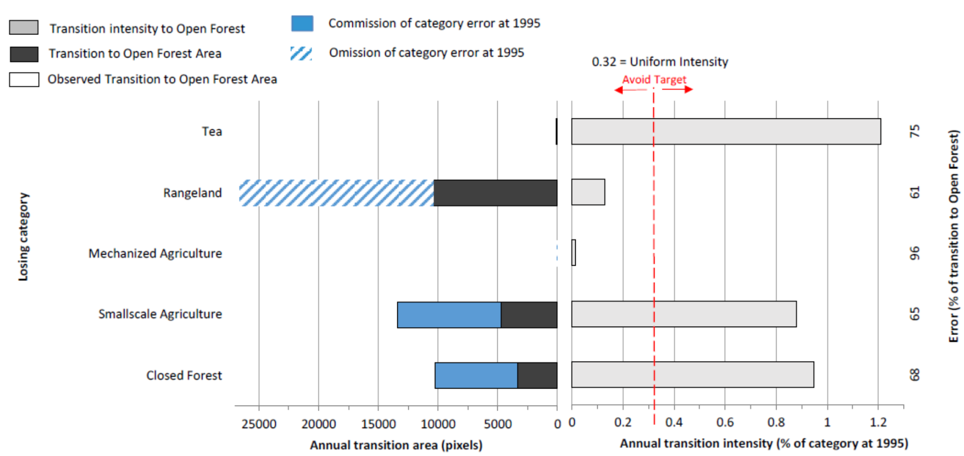

Figure 3.6: Transition to open forest (1995-2003 interval).

Table 3.7: Land use transitions between categories (focus: deforestation)

	1976-1985		1985-1995		1995-2003		2003-2014	
a) Transition from closed forest *(TI values as % of respective categories at the end of respective intervals)*								
	UI=0.22		UI= 0.43		UI = 0.67		UI = 0.45	
(gaining categories)	TI	EI	TI	EI	TI	EI	TI	EI
Open forest	**1.72**	89.17	**0.94**	56.43	**3.48**	85.35	**2.35**	85.02
Small scale agric	**0.67**	69.00	**1.1**	63.23	**0.67**	(0.30)	**0.64**	30.87
Mechanized agric	0.15	(31.96)	0.03	(93.86)	0.19	(72.56)	0.32	(29.84)
Rangeland	0.06	(73.17)	0.31	(28.91)	0.55	(19.26)	0.3	(34.33)
Tea			**2.33**	85.06	**0.7**	4.32	0.4	(12.13)
b) Transition from open forest *(TI values as % of respective categories at the end of respective intervals)*								
	UI=1.11		UI= 0.46		UI = 0.38		UI = 0.19	
(gaining categories)	TI	EI	TI	EI	TI	EI	TI	EI
Closed forest	**2.66**	64.75	**0.94**	53.41	0.11	(71.48)	**0.64**	71.97
Small scale agric	**3.14**	71.83	**2.69**	86.87	**1.2**	70.81	**0.37**	50.65
Mechanized agric	**4.33**	(82.62)	0.33	(30.38)	0.31	(18.54)	0.15	(21.76)
Rangeland	0.70	(39.80)	0.02	(96.63)	0.15	(60.08)	0.09	(53.84)
Tea			**3.36**	90.44	**5.68**	96.28	**1.65**	90.52
c) Transition to open forest *(TI values as % of respective categories at the start of respective intervals)*								
	UI= 0.26		UI= 0.20		UI = 0.32		UI = 0.17	
(losing categories)	TI	EI	TI	EI	TI	EI	TI	EI
Closed forest	**1.32**	82.11	**0.3**	35.53	**0.95**	67.56	**0.69**	76.59
Small scale agric	**1.4**	83.29	**0.95**	80.83	**0.88**	64.87	**0.27**	38.06
Mechanized agric	0.0	(100)	**1.24**	85.85	0.01	(96.13)	0.14	(18.87)
Rangeland	0.06	(78.24)	0.12	(41.95)	0.13	(60.94)	0.06	(63.71)
Tea					**1.21**	75.21	**0.61**	73.09
d) Transition to small scale agriculture *(TI values as % of respective categories at the start of respective intervals)*								
	UI= 0.59		UI= 0.89		UI = 1.32		UI = 0.64	
(losing categories)	TI	EI	TI	EI	TI	EI	TI	EI
Closed forest	**0.76**	23.9	**1.61**	49.05	**1.34**	1.88	**1.51**	61.92
Open forest	**2**	74.53	**6.48**	94.69	**7.68**	92.61	**2.81**	83.06
Mechanized agric	0	(100)	**1.28**	33.17	**3.87**	73.7	**1.77**	68.6
Rangeland	0.32	(46.51	0.43	(54.04)	1.01	(25.48)	0.39	(41.35)
Tea					**3.93**	74.3	**4.59**	92.56

UI = Uniform intensity (% of the size of other categories, excluding the concerned category); TI = Transition Intensity; EI = Error intensity (% of transition). EI values in brackets are Omission intensity; EI values without brackets are Commission Intensities. (Bold TI values indicate that the land use category is targetted (for gain or loss) by the land use change involved, non-bold TI values indicate the land use category is avoided by the concerned change).

3.3.3 Forest recovery

There is a systematic targeting transition from open forest to closed forest during the last interval (2003-2014) (Figures 3.7 and 3.8). This may be a signal of forest recovery, albeit at a slow rate of about 3km^2/year. This indicates that the recent government effort of evicting people from the forest reserve (NEMA, 2013; Boone, 2012; Nkako et al., 2005) may be working. The slow rate of recovery may be as a result of the fact that the eviction efforts have not been fully successful in some parts where some evicted communities still find their way back to the forest reserve. The slow rate of recovery may also be caused by the fact that the government is using natural regeneration method of forest recovery which may be quite slow (Mullah et al., 2012).

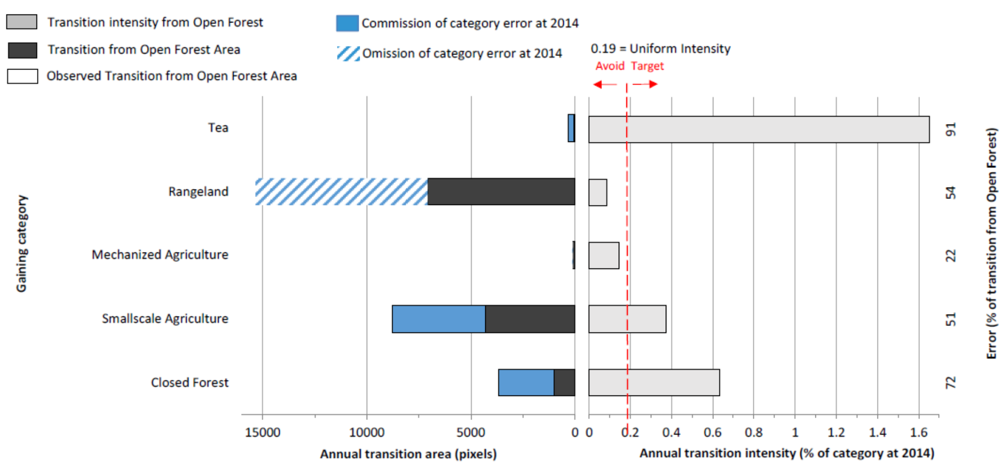

Figure 3.7: Transition from open forest.

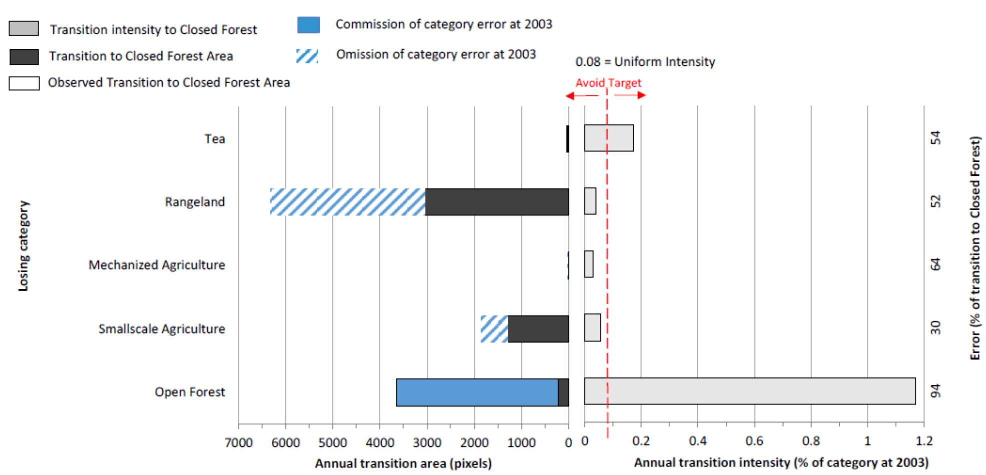

Figure 3.8: Transition to closed forest

3.3.4 Expansion of mechanized agriculture in the rangeland

The middle catchment is mainly occupied by rangelands which is the largest land use category in the basin. Results show that the rangelands have been continuously targeted for expansion of mechanized agriculture (Table 3.8). The conversion was however more intensive in the middle two intervals (1985-1995 and 1995-2003) where the transition from rangelands systematically targeted mechanized agriculture i.e. loss of rangeland most intensively targeted gain of mechanized agriculture (Table 3.8a) and gain of mechanized agriculture targeted loss of rangelands (Table 3.8b). This systematic transition which is stationary in the two intervals shows a strong signal of intensive expansion of mechanized agriculture in the rangeland. It is also worth noting that the annual rate of loss of rangeland was fastest during this period i.e. 1995-2003 interval (Table 3.5), which indicates that other than deforestation and expansion of small scale agriculture, conversion of rangeland to mechanized agriculture also contributed to relatively fast change observed between 1995 and 2003.

This systematic process of intensive expansion of mechanized agriculture into rangelands during this period (1985-2003) can be attributed to change of land tenure of the pastoral land surrounding the Maasai Mara National Reserve. This land is mainly owned by the Maasai community who are pastoralists. The Maasai community has been historically using this land (all the way from Mau forest down to Tanzania) communally for grazing even before 20th century (Waller, 1990). During the colonial period, the British colonial government created ‘Trustlands’ through the Trust Land Act, of 1939 (Wayumba, 2004). Trustland is a communal land tenure system where the land is held in trust by the county councils on behalf of the residents of the county councils jurisdiction (Wayumba, 2004; Nkako et al., 2005). In order to organize and develop livestock production in pastoral lands, the colonial government established large extended family (clan) grazing schemes aimed at solving the problem of overstocking and overgrazing in the Northern and Southern Maasai reserves, created previously to contain the movement of the Maasai (Ng’ethe, 1992; Hughes, 2007). It is upon this concept of grazing schemes that the Government of Kenya conceived the idea of ‘group ranches’ land tenure model after independence in 1963 (Ng’ethe, 1992; ole Sandera, 1986), which was embedded in law by Land (group representative) Act of 1968 (GoK, 1968). In the areas where group ranches were created, the existing ‘Trustland’ land tenure system was changed to the group ranch land tenure system (Davis, 1970). The group ranches land tenure system is a communal land ownership system where land is held in trust by a few selected people (3 to 10) on behalf of the members of the group ranch (Wanyumba, 2004). The membership of the groups was based on kinship i.e. members of the tribe, clan, family or any other group of people who owned communal land recognized under customary law (GoK, 1968; Kimani and Pickard, 1998).

From late 1970s and early 1980s, the Maasai were agitating for subdivision of group ranches mainly due to inefficient management by the elected committee members as well as the pressure to include, into group membership, other youthful members who were young during group formations and had come of age (Kimani and Pickard 1998; Thompson and Homewood, 2002). Consistent push by the Maasai prompted the government to accept the subdivision of the land in 1983. Members of the subdivided group ranch received their share of parcel of land with private title deed (Kimani and Pickard 1998; Wayumba, 2004). This subdivision and privatization of the initially communal land to private land has since brought about major changes in land use in the area; mainly the expansion of mechanized agriculture (Thompson and Homewood, 2002).

It is therefore obvious that the intensive expansion of the large scale mechanized agriculture in the rangelands observed during the two intervals (1985-1995 and 1995-2003) coincided with the subdivision and privatization of the group ranches. Lemek group ranch (formed in 1969), for example, was sub-divided between 1993 and 1999; Maji Moto was subdivided in 1999; Koiyaki was subdivided by 2003, while Ol Kinyei finished their subdivision by 2004/2005 (Snider, 2012; Thompson et al., 2009; Zeppel, 2006; Thompson and Homewood, 2002).

Narok is a high potential wheat growing area (Jaetzold and Smith, 1983). Farmers, particularly from agricultural communities in Kenya, lease big tracks of land for large scale wheat farming. Even prior to subdivision of the group ranches, these commercial farming entrepreneurs negotiated concessions with group ranch committee members on behalf of the other members. However, mismanagement of income from the leases also partly contributed to members agitating for subdivision of the group ranches (Thompson and Homewood, 2002). After subdivision, the individual members were able to directly lease out their land for commercial farming. This caused the intensive expansion of mechanized agriculture as observed in our study. By 1995, for example, about 24% of the 495km^2 Lemek group ranch was converted into mechanized agriculture (Thompson and Homewood, 2002). During the same period, Thompson and Homewood (2002) reported expansion of small scale farming in the rangeland by the Maasai but mainly for subsistence. The study by Thompson and Homewood (2002) estimated that about 53% of the households in the former group ranches were practicing small scale agriculture outside their homesteads. The Maasai being pastoralists, only small areas (less than 2 acres) were used for cultivation surrounded by land used for grazing.

Table 3.8: Land use transitions between categories (Focus: expansion of agriculture)

	1976-1985		1985-1995		1995-2003		2003-2014	
a) Transition from rangelands *(TI values as % of respective categories at the end of respective intervals)*								
	UI= 1.96		UI= 2.22		UI = 4.14		UI = 1.42	
(gaining categories)	TI	EI	TI	EI	TI	EI	TI	EI
Closed forest	1.78	(10.82)	**2.62**	19.72	**4.37**	7.56	0.53	(66.53)
Open forest	0.77	(65.32)	**2.93**	31.06	3.53	(20.61)	**1.67**	17.68
Small scale agric	**2.88**	38.82	**2.39**	9.01	3.75	(13.45)	1.28	(11.91)
Mechanized agric	**6.31**	83.71	**8.11**	93.35	**7.98**	71.91	**2.7**	56.1
Tea			**3.26**	41.03	0.79	(86.40)	0.32	(80.1)
b) Transition to mechanized agriculture *(TI values as % of respective categories at the start of respective intervals)*								
	UI= 0.02		UI= 0.01		UI = 0.026		UI = 0.04	
(losing categories)	TI	EI	TI	EI	TI	EI	TI	EI
Closed forest	0	(81.62)	0	(96.72)	0.01	(79.68)	0.02	(48.61)
Open forest	**0.05**	66.91	0.01	(36.73)	**0.03**	4.41	0.03	(24.59)
Small scale agric	0.01	(54.79)	**0.02**	15.68	0.02	(29.20)	**0.11**	61.07
Rangeland	0.01	(22.72)	**0.02**	15.19	**0.03**	12.15	0.02	(44.62)
Tea					0	(100)	0.03	(22.12)

UI = Uniform intensity (%of the size of other categories, excluding the concerned category)); TI = Transition Intensity; EI = Error intensity (% of transition). EI values in brackets are Omission intensity; EI values without brackets are Commission Intensities

3.3.5 Effect of wildlife associations and conservancies

During the last interval, expansion of mechanized farming has avoided the rangelands and targeted small scale agriculture (Figures 3.9 and 3.10) in a systematic transition. This indicates a shift of expansion of mechanized farming from rangelands to land under small scale agriculture near the Maasai Mara National Reserve. Norton-Griffiths et al. (2008) observed a shift from mechanized farming predominantly on large plots leased to large scale contractors prior to 2004 to an increased mechanized farming based on smallholder enterprises, often farmed by Maasai owners themselves. Similar observations were made by Thompson et al. (2009) who reported that Maasai farmers were increasingly managing medium sizes mechanized cultivation plots using tractors. This may have been as a result of formation of wildlife associations and conservancies in recent years. The group ranches surrounding the Maasai Mara National Reserve were (and still are) used as wildlife dispersal areas. After subdivision of the group ranches, tour operators and hoteliers made arrangements with individual landowners around the Masaai Mara National Reserve where landowners would stop leasing out their land for cultivation and instead use it as wildlife dispersal areas. The landowners would in turn get some revenue from the tour operators who took tourists for game drive as well as hoteliers who set up camps for tourists. As a result, Maasai farmers themselves aggregated the small farms plots they were using for subsistence farming for mechanized (medium-scale) cultivation and left the other land for wildlife dispersal. For example, some members of the Koiyaki and Lemek group ranches came together and formed a community based conservation project established in 2001 (Manyara and Jones, 2007). The Koiyaki-Lemek conservation Trust charged game viewing fees and had contract with 25 tour operators that leased their campsites on Maasailand (Zeppel, 2006).

In the last decade, some Maasai land owners came up together and started wildlife conservancies on their land outside the Maasai Mara National Reserve (Sørlie, 2008). In this new arrangement, the private land owners adjacent to the MMNR pool their land together to create a big game viewing area viable as a conservancy and then broker lease agreements with tour operators under Payment for Ecosystem Services (PES) model (Osano et al., 2013). The lease agreements range between 5 to 15 years (Homewood et al., 2012). In the conservancies, the landowner who join the PES schemes agree to move out and are not allowed to sell their land, construct homesteads, cultivate, fence or graze their animals (Homewood et al., 2012). Therefore, these wildlife conservation efforts in the Mara may have seen the shift of mechanized cultivation from targeting rangelands to the areas that were already under small scale cultivation in the rangeland.

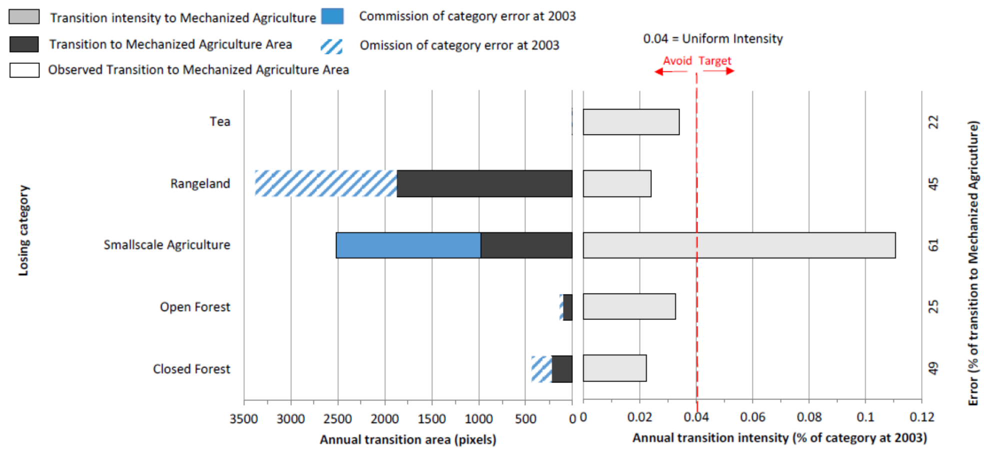

Figure 3.9: Transition to mechanized agriculture (2003-2014).

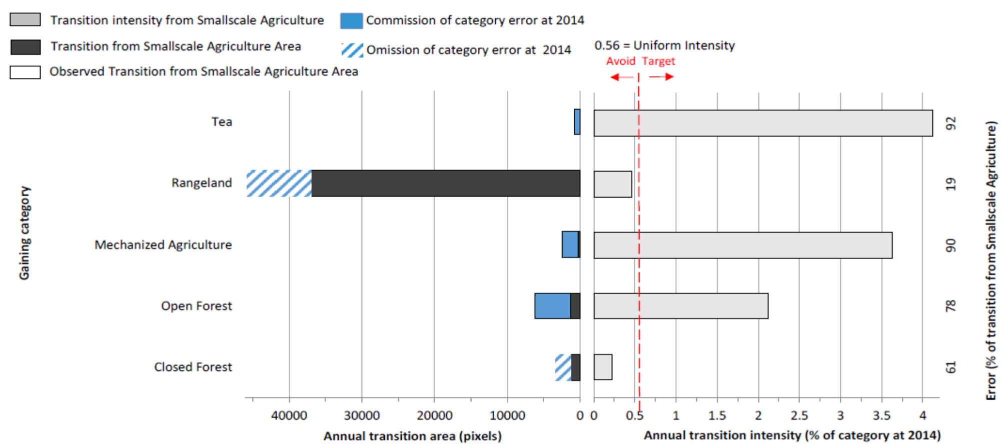

Figure 3.10: Transition from small scale agriculture (2003-2014 interval).

3.3.6 Impact of land use change

The observed land use changes in the Mara River basin, mainly deforestation and expansion of agriculture both in the formerly forested areas of the Mau Forest Complex and in the rangelands, are bound to have some effects on the watersheds capacity to provide ecosystem services. Some studies have found that deforestation and expansion of agriculture has affected the hydrology of the watershed (Mati et al., 2008; Mwangi et al., 2016b; Kiragu, 2009). Mati et al. (2008) found that land use change between 1973 and 2000, had increased the peak flow of Mara River by 7%. Mwangi et al. (2016b) estimated that land use change in the last 50 years contributed to 97% of the observed increase in mean streamflow of Nyangores River (a headwater tributary of the Mara

River). Kiragu (2009) observed increased sedimentation of the Mara River which he attributed to deforestation and intensive agriculture currently practiced in the watershed. The increase in mean streamflow observed by Mwangi et al. (2016b) was attributed to reduced water use by vegetation (transpiration) following deforestation. Trees are generally known to consume (transpire) more water than most vegetation and therefore deforestation reduces water removal by trees from soil and groundwater. The excess groundwater in shallow aquifers is availed as baseflow component of streamflow. Deforestation and intensification of agriculture are likely to cause increase in surface runoff due to degradation of the watershed which reduces its capacity to absorb rainwater (reduced infiltration) (Recha et al., 2012). This may manifest as increased peak flows as observed by Mati et al. (2008) in the Mara River Basin. Increased surface runoff, especially on agricultural land, accelerates soil erosion and subsequent sedimentation in rivers, as observed by Kiragu (2009) in the case of MRB.

Ogutu et al. (2009) attributed the decline in some wild animal species (e.g. giraffe, waterbuck and impala) to land use change in the pastoral ranches bordering the Maasai Mara National Reserve, mainly caused by progressive habitat deterioration. Mau forest has a rich biodiversity of plants (including indigenous trees), birds and wild animals (Kinyanjui et al., 2014; GoK, 2009). However, deforestation is destroying critical habitats for some species that are of conservation significance such as *Cisticola Aberdare* bird, yellow backed and Blue Duikers, and Giant forest hog (GoK, 2009; Nkako et al., 2005; Muiruri and Maundu 2010; BirdLife International 2016). Mau forest has also been a home to Ogiek (Dorobo) people who are a hunter-gather community traditionally occupying the moist montane forest areas in Kenya, especially along the Mau escarpment (Klopp and Sang, 2011; GoK, 2009). Deforestation of the Mau forest and occupation by agricultural communities has resulted in ethnic clashes in the area.

It is therefore obvious that the observed land use change in the Mara River Basin has affected the watershed’s capacity to provide some ecosystem services (e.g. good quality water, good habitat for some wild animals and people). It is important therefore that the patterns, trends and dominant processes of land use change (deforestation and expansion of agriculture) observed in this study be used to develop strategies to arrest further deforestation of the natural forest and sustainably control expansion of agriculture.

3.4 Conclusions

This study uses intensity analysis approach to examine the patterns, dynamics and processes of land use change within Mara River basin in East Africa in four consecutive time intervals between 1976 and 2014. We mainly focused on transitions among forest, agriculture and rangelands to reveal more details on deforestation and expansion of agriculture in the basin. The overall land use/cover change was fastest during the 1995-2003 interval and have slowed down during the last interval (2003-2014). Swap changes accounted for more than 50% of the total change in the intervals, which indicates that the land use/cover change is very dynamic i.e. accompanied with high relocation of land use categories within the watershed.

A systematic transition between loss of closed forest and gain of open forest was observed (i.e. gain open forest intensively targeted closed forest for takeover). This transition was stationary over the entire study period (same across all the intervals). The stationary systematic transition implies that loss of closed forest to open forest is a dominant (prominent) land use change within the basin. A stationary systematic transition between loss of open forest and gain in small scale agriculture was also observed. This implies that open forest has consistently been losing to small scale agriculture throughout the entire study period, which is another dominant land use change in the watershed. We attributed the observed deforestation to continuous encroachment and a series of excisions of the forest reserves particularly the Mau forest complex at the headwaters of the Mara River. The large scale excision of 2001 partly contributed to land use/cover change being fastest during the 1995-2003 interval. These two systematic transitions (i.e. from closed forest to open forest and from open forest to small scale agriculture) also show a trend (pathway) of deforestation from closed forest to small scale agriculture, with open forest as a transitional land cover. This trend implies that closed forests are first opened-up (probably for timber and charcoal) and then the opened patches are cultivated. Eventually, remaining trees are removed (logged) as cultivation expands into the open forest. During the last interval (2003-2014) a systematic transition between loss of open forest and gain of closed forest was observed, which indicate that recent government effort to evict people illegally occupying the forest may be working, albeit at a slow rate. Further studies are required to investigate the whole process of encroachment into the forest reserves e.g. who (locals or outsiders) and for what purpose (e.g. timber or charcoal) is the closed forest opened up?; how long it takes for cultivation to start in the opened up forest?; what are the reasons behind some parts of opened forest being cultivated and settled into while others experiences regrowth?

Another systematic transition was observed between loss of rangeland and gain of mechanized agriculture during the two middle intervals (1985-1995 and 1995-2003). This implies than rangeland was intensively losing to mechanized agriculture between 1985 and 2003. This was attributed to change of land tenure (from communal to private) especially in the rangeland. Most of group ranches (under communal land tenure) in the area were subdivided to members during this period which may have accelerated leasing of land to mechanized large scale farmers. The fastest interval of overall land use change (1995-2003) also fall in this period (1985-2003) which implies that conversion of rangeland to agriculture also contributed to fast change in land use/cover in this interval, in addition to deforestation. Between 2003 and 2014, expansion of mechanized agriculture has avoided gaining from rangelands to systematically targeting gaining from small scale agriculture. This implies small plots used for smallholder agriculture in the rangeland are coalescing to bigger plots for mechanized cultivation. This was attributed to recent establishment of wildlife associations and conservancies where landowners make agreement with tour operators and hoteliers to stop leasing their land for cultivation and use it as wildlife dispersal areas, with some monetary incentives in return.

Further studies on the possible driving forces of deforestation and expansion of agriculture identified in this study may be required for development of effective management strategies. It may be important, for example, to determine the influence of political patronage and forest management institutions on excisions forest reserves and how these excisions impact further encroachment of the forest reserves. It may also be important to investigate the influence of political landscape and flow of income from wildlife conservation (from both Maasai Mara National Reserve and from wildlife associations and conservancies) affect conversion of rangelands to agriculture and vice versa.

4 Chapter four: Relative contribution of land use change and climate variability on discharge of upper Mara River, Kenya

Publication (this chapter has been published in Journal of Hydrology: Regional studies)

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Abstract

Study Region

Nyangores River watershed, headwater catchment of Mara River basin in Kenya

Study Focus

Climate variability and human activities are the main drivers of change of watershed hydrology. The contribution of climate variability and land use change to change in streamflow of Nyangores River, was investigated. Mann Kendall and sequential Mann Kendall tests were used to investigate the presence and breakpoint of a trend in discharge data (1965-2007) respectively. The Budyko framework was used to separate the respective contribution of drivers to change in discharge. Future response of the watershed to climate change was predicted using the runoff sensitivity equation developed.

New Hydrological Insights for the Region

There was a significant increasing trend in the discharge with a breakpoint in 1977. Land use change was found to be the main driver of change in discharge accounting for 97.5% of the change. Climate variability only caused a net increase of the remaining 2.5% of the change; which was caused by counter impacts on discharge of increase in rainfall (increased discharge by 24%) and increase in potential evapotranspiration (decreased discharge by 21.5%). Climate change was predicted to cause a moderate 16% and 15% increase in streamflow in the next 20 and 50 years respectively. Change in discharge was specifically attributed to deforestation at the headwaters of the watershed.

4.1 Introduction

Changes in watershed hydrology may have far reaching impacts on a catchment water balance. The changes may be observed through change in water input (precipitation), water distribution into evapotranspiration and runoff, and in the short term, change in catchment water storage (i.e., soil storage and groundwater recharge). Climate variability and human activities are the main drivers of changes in watershed hydrology (Tomer and Shilling, 2009; Ye et al., 2013). At a local scale, change in precipitation may only be caused by changes in climate, while changes in streamflow, evapotranspiration and watershed storage may be caused either by climate variability, human activities or both. Changes in streamflow (either total water yield or seasonal discharge) have a major implication on water resources management and especially water supply (Döll and Schmied, 2012; Farley et al., 2011; Charlton and Arnell, 2011). Human activities can alter streamflow through changes in land use, reservoir operation and direct abstraction of surface water or groundwater (Carpenter et al., 2011; Biemans et al., 2011). In absence of reservoirs and inconsiderable water abstractions, land use change and climate variability are the main drivers of change in streamflow (Carpenter et al., 2011). Separation of the impacts of the drivers is helpful in better understanding of the watershed hydrology as well as in developing sound water resources management strategies (DeFries and Eshleman, 2004; Arnell and Delaney, 2006). However, separation and quantification of the drivers' impact is challenging (Zhang et al., 2014; Li et al., 2009; Tomer and Shilling, 2009) because of the complex linkage between climate, human activities and the individual hydrological processes (Falkenmark and Rockström, 2004).

A number of studies have proposed approaches to separate the impacts of land use change and climate variability on streamflow (Li et al., 2012; Wang, 2014). The approaches can be broadly categorized as empirically-based and process-based. Proposed empirical methods are based on climate elasticity (Schaake, 1990) and test the sensitivity of streamflow to changes in climatic factors (Ma et al., 2010). Elasticity-based methods can further be categorized into non-parametric and water balance based methods (Sun et al., 2014). Non-parametric elasticity-based methods are empirical approaches that use linear relationships derived from long-term historical data (Schaake, 1990; Sankarasubramanian et al., 2001; Zheng et al., 2009; Ma et al., 2010). Most of the water balance-based elasticity methods (Dooge et al., 1999; Arora, 2002; Wang and Hejazi, 2011; Roderick and Farquhar, 2011) are based on the concept of the Budyko framework

(Budyko, 1974) of catchment water-energy budget (Sun et al., 2014). Process-based methods use distributed physically-based hydrological models where separation is done by alternatively varying and fixing (holding constant) the meteorological inputs and land use/cover conditions (Xu et al., 2014). Process-based methods are more sophisticated, require more data as input and have high uncertainty in parameter estimation whereas non-parametric elasticity methods have weak or no physical meaning (Xu et al 2014; Wang and Hejazi, 2011). Approaches based on catchment water-energy budgets are easier to use and also have better physical background (Sun et al 2014; Roderick and Farquhar, 2011).

In this study, we used the catchment water-energy budget approach to separate the contribution of climate variability and land use change on discharge of Nyangores River; the river is a tributary of the trans-boundary Mara River in East Africa. Over the watershed of Mara River, competing land uses and socio-economic activities in the headwaters have been blamed for changes in its hydrological regime (Gereta et al., 2009; Mati et al., 2005, 2008; Dessu and Melesse, 2012). There has been significant deforestation and conversion to agriculture in the upstream regions of the Mara River basin (Mutie et al., 2006). Other studies have also linked observed high level of sediment yield and sedimentation in the Mara River to land degradation following deforestation (Kiragu, 2009; Defersha and Melesse, 2012). A land use change analysis study by Mati et al. (2008) found that the forest cover of 1973 in the Mara basin progressively decreased by 11% and 32% in 1986 and 2000 respectively. For the same periods, open forest increased by 73% and 213% respectively based on the 1973 land cover - a clear indication of the massive deforestation that was taking place in the area immediately after Kenya's independence in 1963. At independence, almost the entire upstream area of the Mara River basin including the Nyangores watershed was covered by dense natural forest and pockets of montane grassland (Government of Kenya - GoK, 1969). Cultivation was limited and strictly controlled by the colonial government (Kanogo, 1987). Mati et al. (2008) used the 1973 and 2000 land use maps to simulate the effect of land use change on hydrology of the Mara River. They found an increase in peak flow during the long rainfall season (March-May) between 1973 and 2000 which they attributed to deforestation in the basin. Mango et al. (2011) simulated deforestation in Nyangores watershed and likewise reported that further deforestation in the watershed may increase peak flows and reduce dry season flows. Based on the findings of these two studies, it can be deduced that deforestation (past or future) lead to increase in peak flows in the watershed. Change in streamflow, however, is not only caused by human activities (particularly land use change) but also by climate variability. Information on how much of observed change in streamflow is separately caused by land use change and climate variability is important for water resources management planning including simulation of informed future land use and climate change scenarios. Analysis of measured historical streamflow data gives valuable evidence-based information of watershed response to past changes in land use and climate variability either individually or in combination. Such information is however lacking for the Mara River basin.

Separation of the contribution from drivers of change in observed streamflow i.e. land use and climate variability is important for integrated watershed management in the Mara River basin. Herein, we focus on Nyangores watershed, one of the headwater catchments of Mara River basin where there has been a major competition between forest conservation and agriculture. The objectives of the study are: (i) to statistically test the presence of a trend in measured streamflow data, (ii) to empirically separate hydrological impacts caused by changes in land use and climate variability from historical streamflow data, (iii) to further partition the contribution of climate variability into that caused by changes in rainfall and potential evapotranspiration respectively, and (iv) to predict the future relative contribution of climate change to streamflow.

4.2 Materials and methods

4.2.1 The study area

The Mara River has a unique watershed that is characterized by several spatially-varied land uses: forest conservation and smallholder agriculture in the headwaters, wildlife conservation, pastoralism and large-scale agriculture in the mid-catchment, and mining and smallholder agriculture downstream. The watershed, therefore, is a major contributor to the economy of the region, especially through the wildlife-based tourism in the two national game reserves the watershed hosts (i.e., the Maasai Mara National Reserve and the Serengeti National Reserve). The headwater catchments (Nyangores and Amala) are the lifeline of the Mara River especially in dry weather season when they contribute more than 50% of streamflow (McClain et al., 2014; Dessu et al., 2014).

Nyangores River is a tributary of the Mara River which originates from Mau Forest in Kenya, flows through the Masaai Mara and Serengeti National Reserves in Kenya and Tanzania respectively and finally drains into Lake Victoria (Figure 4.1a). Nyangores watershed covers an area of 690 km^2 and is located in the upper part of the trans-boundary Mara River basin (Figure 4.1a). Lying at an altitude range of 1900 – 2970 m above sea level, the watershed main land uses are forest (Mau) and (cropland) agriculture. The main soils are Andosols and Nitisols (World Reference Base –Food and Agriculture Organization of the United Nations classification). The region receives bimodal rainfall pattern with long rains between March and May, and short rains between October and November. The mean annual rainfall is about 1370 mm.

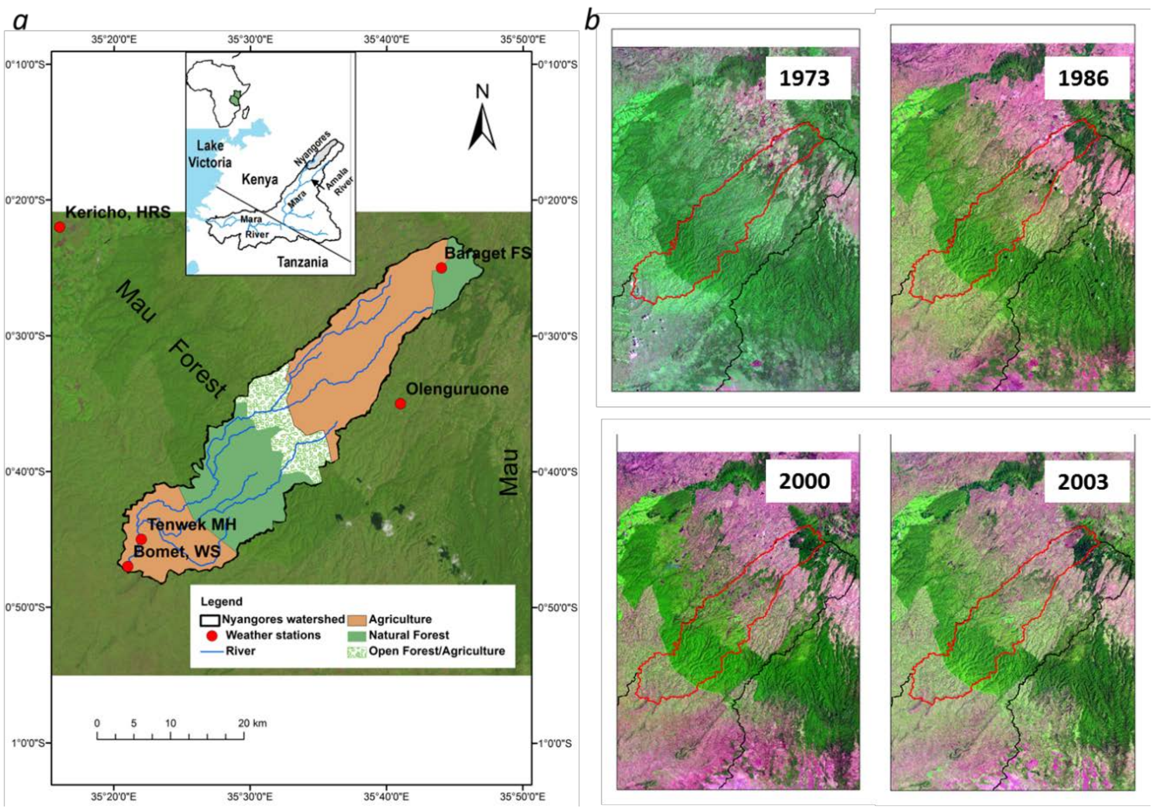

Figure 4.1: (a) Nyagores River watershed; (b) Landsat satellite images showing the forest decline in Nyangores watershed. Dark green show the natural forest; light (faded) green and pink show cleared forest and cultivated land respectively.

4.2.2 Data

Daily discharge of River Nyangores recorded over the period 1965-2007 from the gauging station (1LA03) at Bomet town was granted for this study by Kenya Water Resources

Management Authority.The meteorological data was obtained from Kenya Meteorological Department. Daily rainfall data was obtained for Bomet water supply, Tenwek mission hospital, Olenguruone District Officer's office and Baraget forest stations (Table 4.1; Figure 4.1a). Monthly average data for temperature (T_{max}, T_{min}) (Figure 4.3), wind speed, solar radiation and relative humidity was obtained for Kericho Hail research station (Figure 4.1a). Potential evapotranspiration (PET) (Figure 4.3) was calculated using Food and Agriculture Organization of United Nations (FAO) Penman-Monteith method (Allen et al., 1998). Several methods for estimation of PET are available in literature, some based on temperature (e.g. Hargreaves and Thornthwaite) and others based on radiation (e.g. Priestley-Taylor) (Tegos et al., 2015; Lu et al., 2005). FAO Penmann-Monteith method is a hybrid method that incorporates all climatic and biological factors affecting evapotranspiration. It has been widely applied in range of climatic conditions and found to give better estimates of PET compared to other methods (Garcia et al., 2004; Cai et al., 2007; Gavilán et al., 2006; Jabloun and Sahli, 2008; Ngongondo et al., 2013; Tegos et al., 2015). Though it requires more climatic data than most of the other methods, Allen et al. (1998) outlined a procedure for estimation of PET using FAO Penman-Monteith equation with limited data thus making it applicable in a wide range of conditions (Jabloun and Sahli, 2008; Garcia et al., 2004). Short gaps in the daily discharge data (ca. 5 days) were filled using linear interpolation and inference method using the hydrograph of the adjacent topographically similar Amala River watershed (Figure 4.1a) (Rees, 2008); years with long continuous gaps (e.g., 1993-1995) were excluded from the time series analyses. Missing daily rainfall data was filled by arithmetic mean of rainfall recorded for the particular day in the neighbouring stations. Average annual areal rainfall for the watershed was estimated by Thiessen polygon method (Szcześniak and Piniewski, 2015; Thiessen, 1911). The daily streamflow data was aggregated into mean annual discharge and was expressed as depth (mm) using Equation (4.1) so as to conform to the units (mm) of rainfall (Figure 4.2) and PET.

$$Discharge\left(\frac{mm}{day}\right) = \frac{Discharge\ (m^3/s) * (3600 * 24)}{Watershed\ area\ (m^2) * 1000} \tag{4.1}$$

Table 4.1: Overview of rainfall data

Station Name	station ID	From	To	% complete	Annual mean (mm)	SD*	CV*
Bomet Water Supply	9035265	1967	2009	88	1363	226	0.17
Olenguruone District officer Office	9035085	1960	2002	83	1520	406	0.27
Baraget Forest Station	9035241	1961	1998	95	1138	235	0.21
Tenwek Mission Hospital	9035079	1960	2008	93	1448	172	0.12

*SD is the standard deviation; CV is the coefficient of variation (SD/Mean)

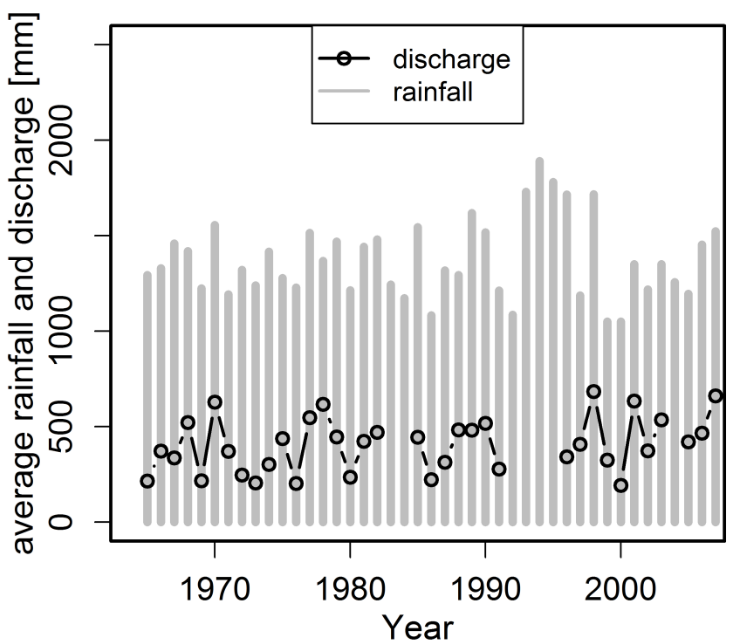

Figure 4.2: Annual discharge for Nyangores River (at Bomet town gauging station) and average annual rainfall.

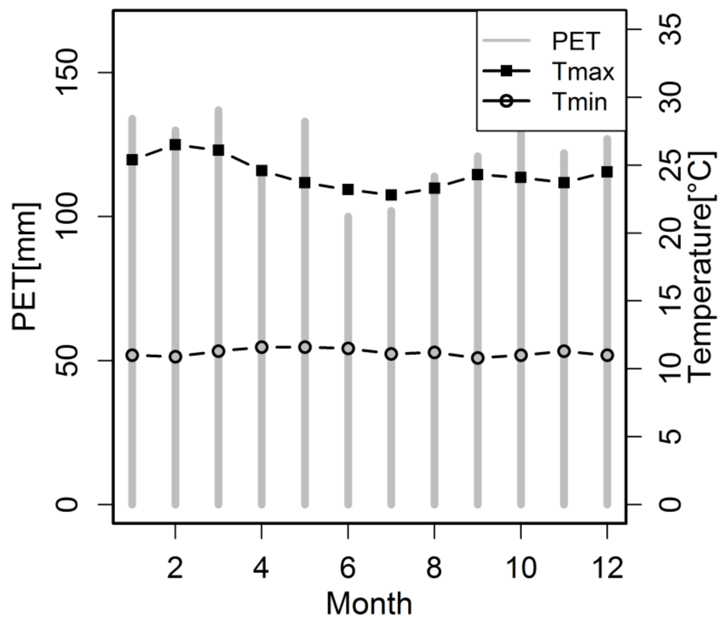

Figure 4.3: Monthly temperature (maximum and minimum) and potential evapotranspiration.

4.2.3 Trend analysis and breakpoint test

4.2.3.1 *Mann Kendall test*

The Mann Kendall test (Mann, 1945; Kendall, 1975) was used for trend analysis of the streamflow data. The method has been widely used for trend analyses in hydro-climatic studies (e.g., Zhang et al., 2015; Ye et al., 2013; Ongoma et al., 2013; Xu et al., 2014; Sun et al., 2014). This test is a rank based non-parametric method used for change detection in a time series. It accommodates missing values and outliers, and data with skewed distributions (Partal and Kahya, 2006; Hirsch and Slack, 1984). However, it has been shown that the results of the original version of Mann Kendall method are affected by serial correlations (von Storch, 1995) which may increase the probability of detecting trends when they don't exist and vice versa (Yue et al., 2002; Hamed and Rao, 1998). Several modifications of the Mann Kendall method have been proposed to limit the influence of autocorrelation in trend analysis of hydro-climatological data (e.g. von Storch, 1995; Hamed and Rao 1998; Yue et al., 2002; Yue and Wang, 2004; Hamed, 2009). The modifications mainly involve prewhitening (transformation of an autocorrelated series into an uncorrelated one before trend test) or modification of variance (Hamed, 2009; Yue et al., 2002). Each of these approaches have associated strengths and weaknesses as shown by several studies (e.g. Sang et al., 2014; Aissia et al., 2014; Zhang and Zwiers, 2004; Yue and Wang, 2002; Yue et al., 2002; Hamed and Rao, 1998) that have explored their robustness in dealing with autocorrelation. In this study, the method proposed by Hamed and Rao (1998) was used. Hamed and Rao (1998) modified the variance of the original Mann Kendall method based on effective sample size. The results were further verified by the method proposed by Yue and Wang (2004) that is also based on effective sample size but computed from the sample serial correlation estimated from a detrended series. The slope of the trend was estimated using Sen's method (Sen, 1968). The Hamed and Rao (1998) method (just like other versions of Mann Kendall) tests a null hypothesis of no trend in the time series. The time series (herein: annual discharge data) is arranged sequentially in order of (the year) measurement. The magnitude of the discharge for each year x_j ($j= 1, 2, \dots n$) is compared with the magnitude of discharge of each of the preceding years x_k ($k = 1, 2, \dots j-1$), ($j > k$). The sign (*sgn*), given by Equation (4.2), is used to count the difference between the two values (x_j and x_k) from the time series.

$$sgn(x_j - x_k) = \begin{cases} 1 & \text{if } x_j > x_k \\ 0 & \text{if } x_j = x_k \\ -1 & \text{if } x_j < x_k \end{cases} \tag{4.2}$$

The test statitsic *S*, which is defined as the total *sgn* of the whole time series is calculated as:

$$S = \sum_{k=1}^{n-1} \sum_{j=k+1}^{n} sgn(x_j - x_k) \tag{4.3}$$

For large series (number of observations, $n \geq 8$), the statistic S is approximately normally distributed with mean and modified variance (Hamed and Rao, 1998) calculated using Equations (4.4) and (4.5) respectively.

$$E(S) = 0 \tag{4.4}$$

$$V^*(S) = V(S).\frac{n}{n_S^*} = \frac{n(n-1)(2n+5) - \sum_{m=1}^{n} t_m m(m-1)(2m+5)}{18}.\frac{n}{n_S^*} \tag{4.5}$$

Where $V(S)$ is the variance of the original Mann Kendall, t_m is the number of data in a tied group (there is a tie when $x_j=x_k$), m is the number of tied groups, n^*_s is the effective sample size and n/n^*_s is the correction factor due to autocorrelation in the data which is calculated as:

$$\frac{n}{n_s^*} = 1 + \frac{2}{n(n-1)(n-2)} \sum_{i=1}^{n-1} (n-1)(n-i-1)(n-i-2)\rho_s(i) \quad (4.6)$$

where n is the actual number of observations and $\rho_s\,(i)$ is the autocorelation function of the ranks of the observations.

The standardized statistic Z follows a standard normal distribution and is given by:

$$Z = \begin{cases} \frac{S-1}{\sqrt{V^*(S)}} & \text{if } S > 0 \\ 0 & \text{if } S = 0 \\ \frac{S+1}{\sqrt{V^*(S)}} & \text{if } S < 0 \end{cases} \quad (4.7)$$

The null hypothesis of no trend is rejected if the absolute value of Z is bigger than the theoretical value of $Z_{(1-\alpha/2)}$ at α level of significance. A positive value of S indicates an upward trend while a negative value indicates a downward trend.

4.2.3.2 *Sequential Mann Kendall test*

The sequential Mann Kendall test (Modarres and Sarhadi, 2009; Sneyers, 1990) was used to detect the occurrence of a breakpoint in discharge. The sequential Mann Kendall test is a graphical technique used to approximate the beginning of a change in a time series based on progressive and retrogressive analysis of the Mann Kendall statistic. Just like in Mann Kendall test, the annual discharge time series is arranged sequentially in order of measurement. The magnitude of the discharge for each year x_j ($j = 1, 2, \ldots n$) is compared with the magnitude of discharge of each of the preceding years x_k ($k = 1, 2, \ldots j-1$), ($j > k$). For each time step (year), the number of cases where $x_j>x_k$ is counted. Then the normally distributed statistic t_j is calculated using Equation (4.8) where n_j denotes the number of cases where, $x_j > x_k$.

$$t_j = \sum_i^j n_j \quad (4.8)$$

The mean and variance of t_j are calculated using Equations (4.9) and (4.10) respectively and then the progressive variable statistic $UF(t_j)$ (forward sequence) is calculated using Equation (4.11). The retrogressive variable statistic $UB(t_j)$ (backward sequence) is calculated with the same Equation (4.11) but with a reversed series of the data.

$$E(t_j) = \frac{j(j-1)}{4} \tag{4.9}$$

$$Var(t_j) = \frac{j(j-1)(2j+5)}{72} \tag{4.10}$$

$$UF(t_j) = \frac{t_j - E(t_j)}{\sqrt{Var(t_j)}} \tag{4.11}$$

The intersection of the forward and the backward curves represented by the graphs of statistics $UF(t_j)$ and $UB(t_j)$ respectively indicates the beginning of the step change point (Partal and Kahya, 2006; Ye et al., 2013; Wang, 2014).

4.2.4 Separating the impacts of land use change and climate variability in runoff

The Budyko framework (Budyko, 1974) was used as the basis to quantify the relative contribution of climate and land use changes to the changes in the watershed hydrology. It is a water and energy balance method that is used to separate the component of precipitation (P) that contribute to evapotranspiration (E) and streamflow (Q). The Budyko hypothesis assumes steady-state water balance conditions of the watershed which require a time scale where change in watershed storage is negligible (e.g., annual basis) (Roderick and Farquhar, 2011). The Budyko curve represents the long-term watershed average evaporative index (i.e., ratio of actual evapotranspiration to precipitation (E/P)) and the aridity index, i.e., ratio of potential evapotranspiration to precipitation (E_0/P) (Donohue et al., 2011). A particular curve has the same catchment property (*n*) at all point along the curve but with different aridity indices (E_0/P) i.e., different climatic conditions (Sun et al., 2014). Thus, the Budyko hypothesis postulates that under stationary watershed conditions, a watershed will fall on Budyko curve while under non-stationary conditions (with effect of land use changes, i.e., change in catchment property -*n*) the watershed will deviate from the curve in a predictable manner. The steady state assumption of Budyko hypothesis requires use of long-term average (at least 1 year) of water balance in a watershed (Roderick and Farquhar, 2011; Donohue et al., 2007; Choudhury, 1999). In this study, the water balance was based on average values (P, E and Q) for time period spanning over 44 years separated into two periods based on the year when the change point in the streamflow time series is identified using the sequential Mann Kendall test. The start of the calendar year coincides with the dry season (January and February) in the watershed thus minimizing the inter-annual change in water storage. The region has minimal 'loss' of water to deep groundwater storage (Dagg and Blackie, 1965; Krhoda, 1988). Water abstraction in the Nyangores River is less than 1% of mean daily discharge (Juston et al., 2014) and there are no significant storage dams on the river (McClain et al., 2014).

This study utilized an empirical model developed by Roderick and Farquhar (2011) to quantify the relative impacts of rainfall, potential evapotranspiration and land use change on change in runoff (discharge). The model is based on empirical Equation (4.12) derived from Budyko hypothesis and proposed by Yang et al. (2008) and Choudhury (1999).

$$E = \frac{PE_0}{(P^n + E_0{}^n)^{1/n}} \tag{4.12}$$

E is the actual evapotranspiration, P is the precipitation, E_0 is the potential evapotranspiration and n is an empirical catchment characteristic that represent catchment properties.

The Roderick and Farquhar, (2011) equation is expressed as:

$$dQ = \left(1 - \frac{\partial E}{\partial P}\right) dP - \frac{\partial E}{\partial E_0} dE_0 - \frac{\partial E}{\partial n} dn \tag{4.13}$$

where

$$\frac{\partial E}{\partial P} = \frac{E}{P}\left(\frac{E_0{}^n}{P^n + E_0{}^n}\right) \tag{4.14}$$

$$\frac{\partial E}{\partial E_0} = \frac{E}{E_0}\left(\frac{P^n}{P^n + E_0{}^n}\right) \tag{4.15}$$

$$\frac{\partial E}{\partial n} = \frac{E}{n}\left(\frac{\ln(P^n + E_0{}^n)}{n} - \frac{(P^n \ln P + E_0{}^n \ln E_0)}{P^n + E_0{}^n}\right) \tag{4.16}$$

dQ, dP, dE_0 and dn are the changes in runoff, precipitation, evapotranspiration and catchment properties respectively.

The differential Equation (4.13) indicates that change in runoff is a function of climate variability and changes in catchment properties. The change in runoff caused by climate variability (dQ^c) is separated to that caused by change in precipitation and that caused by change in potential evapotranspiration. The last term in Equation (4.13) represent the changes in runoff caused by changes in catchment properties. Thus, from Equation (4.13) change in runoff caused by change in climate can be estimated as:

$$dQ^c = \left(1 - \frac{\partial E}{\partial P}\right) dP - \frac{\partial E}{\partial E_0} dE_0 \tag{4.17}$$

Sun et al. (2014) considered the residual change in runoff (dQ^R) to be the difference between the observed change in runoff (dQ_{obs}) and the estimated change in runoff caused by change in climate (dQ^c), and is equivalent to runoff change caused by change in catchment properties (Equation (4.18)). The residual change in runoff also includes short-term change in climate variability (i.e., intra-annual climatic effects such as precipitation intensity and temporal distribution of precipitation and potential evapotranspiration) (Sun et al., 2014; Roderick and Farquhar, 2011). Catchment property *n* cannot be easily measured and its value is usually estimated by fitting it in Equation (4.12) using the observed precipitation, potential evapotranspiration and runoff (Donouhe et al., 2011). Thus, changes in runoff caused by changes in catchment properties can be best estimated by Equation (4.18) (Sun et al., 2014).

$$dQ^R = dQ_{obs} - dQ^c \tag{4.18}$$

Equations (4.17) and (4.18) were used to calculate the changes in runoff caused by changes in precipitation, evapotranspiration and catchment properties. The relative contribution of each was calculated as a percentage of the observed (total) change in runoff.

4.2.5 Runoff sensitivity and prediction of future changes in runoff using IPCC projections

The sensitivity of the runoff to climate variability was estimated using Equation (4.19), also proposed by Roderick and Farquhar (2011). Equation (4.19) predicts the relative change in runoff as a result of unit percent change in precipitation and potential evapotranspiration.

$$\frac{dQ}{Q} = \left[\frac{P}{Q}\left(1 - \frac{\partial E}{\partial P}\right)\right]\frac{dP}{P} - \left[\frac{E_0}{Q}\frac{\partial E}{\partial E_0}\right]\frac{dE_0}{E_0} \tag{4.19}$$

Equation (4.20) was adapted for the watershed, based on Equation (4.19), to predict the sensitivity of runoff to climate change. The equation predicts the expected relative change in runoff based on unit percent change in precipitation, potential evapotranspiration or both.

$$\frac{dQ}{Q} = 2.07\frac{dP}{P} - 1.08\frac{dE_0}{E_0} \tag{4.20}$$

The Intergovernmental Panel on Climate Change (IPCC, 2013a) projected changes in monthly temperature for the region (Table 4.2) were then used to calculate the estimated potential evapotranspiration for the watershed in the near-term (2016 - 2035) and medium-term (2046 - 2065) periods using FAO Penman-Monteith method (Allen et al., 1998). The calculated changes in potential evapotranspiration and IPCC (2013a) projected changes in precipitation were then applied to Equation (4.19) to predict the expected future changes in runoff due to climate change.

Table 4.2: IPCC projected monthly increase* in temperature (0^{C}) for the watershed

Period	2016-2035	2046-2065
Dec – Feb	1	1.5
March – May	1	1.5
June – July	1	2
Sep – Nov	1	1.5

**based on Representative Concentration Pathway (RCP4.5) - median (50%) of the distribution of Coupled Model Inter-comparison Project Phase 5 (CMIP5) - IPCC, 2013a)*

The IPCC fifth assessment report (AR5) (IPCC, 2013b) gives patterns of climate change computed from global climate model output gathered as part of the Coupled Model Inter-comparison Project Phase 5 (CMIP5). The climate change projections are made under the Representative Concentration Pathway (RCP) scenarios which are based on more consistent short-lived gases and land use changes. The scenarios specify emissions and are not based on socio-economic driven (SRES) scenarios used in fourth assessment (AR4) which considered future demographic and economic development, regionalization, energy production and use, technology, agriculture, forestry and land use (IPCC, 2013b). The new scenarios for AR5 are based on Radiative Forcing (RF) which quantifies the change in energy fluxes caused by changes in drivers of climate change. RCP4.5 is one of the four RCP scenarios and aims at stabilization of RF at 4.5 (W/m^2). The values given in Table 4.2 are the estimates of the median (50% percentile) of the mean distribution of the 42 models used in CMIP5. More details about the future IPCC climate change projections can be found in the IPCC fifth assessment report (IPCC, 2013b)

4.3 Results

4.3.1 Changes in measured streamflow

Results from trend analysis of discharge data using the modified Mann Kendall tests (both approaches by Hamed and Rao (1998) and Yue and Wang (2004)) showed an increasing trend (with a slope of 4.75 mm/year) significant at 5% level (Figure 4.4). The change point of the discharge data was identified as the year 1977 (Figure 4.5) using the sequential Mann Kendall test. Based on the identified breakpoint, the precipitation, potential evapotranspiration and discharge data were split into the period before change point (1965 - 1977) and the period after change point (1978 - 2007) as shown in Table 4.3. This Table also shows the average annual values of potential evapotranspiration calculated using FAO Penman-Monteith equation for the two periods respectively. All the three input parameters to the water balance Equation (4.12) were found to have increased between the period before change point and the period after change point. This implies an increase of both the water input (precipitation) and atmospheric demand (potential evapotranspiration) in the catchment. Actual evapotranspiration values were calculated for the two periods as the difference between the averages (averaged over the respective time periods) of measured precipitation and runoff (streamflow) (Table 4.3). Also shown in Table 4.3, are the long-term average annual values of the precipitation, potential evapotranspiration, runoff and the actual evapotranspiration covering the entire period (1965 - 2007) of the study. The long-term values represent the average measured or calculated estimates of the water balance parameters in the catchment.

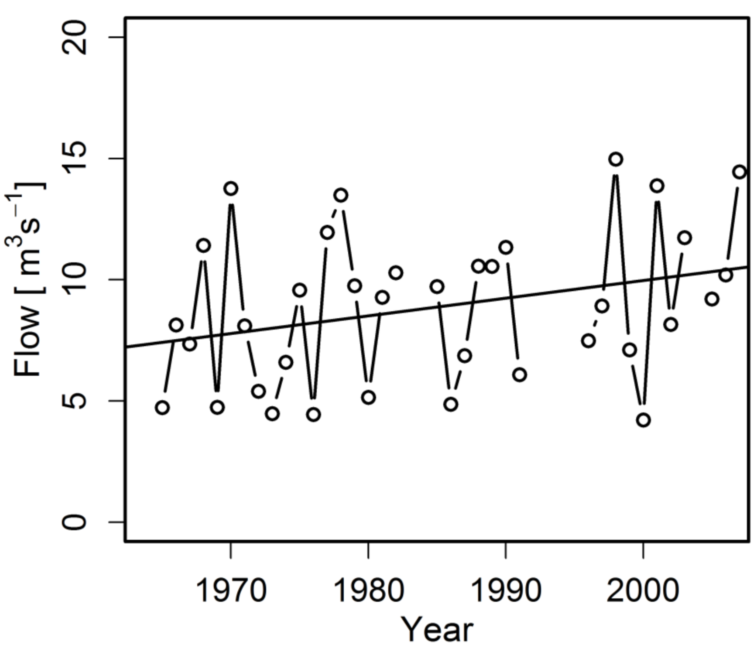

Figure 4.4: Annual discharge of the Nyangores River.

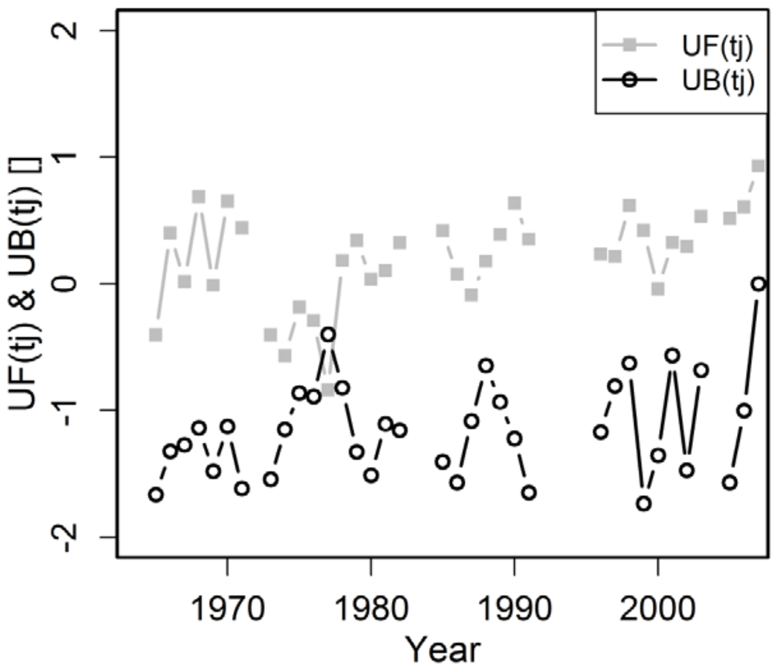

Figure 4.5: Sequential Mann Kendall change point test for discharge data. The intersection of forward sequence statistic UF($_{tj}$) and backward statistic UB($_{tj}$) is the change point in the time series – in this case: 1977.

4.3.2 Catchment properties parameter (n)

The catchment property (n) for the watershed - estimated by fitting it in Equation (4.12) using the long-term mean annual values of precipitation, potential evapotranspiration and streamflow- was found to be 1.75 (Table 4.3). As reflected in Table 4.3 and explained in sections 4.3.3 and 4.4.2, the watershed has undergone through major changes in catchment properties and particularly land use changes.

4.3.3 Hydrological impact of land use change and climate variability

The estimated relative contributions of land use change and climate variability to the observed change in runoff are given in Table 4.4. The results indicate that the observed increase in precipitation (Table 4.3) caused a 24% increase in runoff while on the contrary the estimated increase in potential evapotranspiration caused a 21.6% decline in runoff. Therefore, the net change in runoff caused by the climate variability was only an increase of 2.5%. The rest of the observed change in runoff (dQ^R = 97.5%), denoted as the residual change, was caused by changes in catchment properties which is mainly attributed to land use change as discussed in section 4.4.2. From the results, we conclude that land use change is the main driver of change of the watershed discharge.

Table 4.3: Mean annual values of water balance components (P, Q, E, E_0,) for the period before change point, period after change point and the entire (long-term) period, and catchment parameter (*n*).

	period before change point (1965-1977)	period after change point (1978-2007)	Long-term (1965-2007)
Precipitation (P) (mm)	1342	1382	1373
Potential evapotranspiration (E_0) (mm)	1517	1595	1556
Runoff (Q) (mm)	338	439	405
Actual Evapotranspiration (E = P-Q) (mm)	1004	943	968
Catchment parameter (n)	1.99	1.54	1.75

Table 4.4: contribution of climate variability and land use change to change in streamflow

Driver of change in runoff	Contribution (mm)	Contribution (%)
Precipitation (dQ^p)	+24.4	+24.2
Potential evapotranspiration (dQ^{Eo})	-21.8	-21.6
Climate (dQ^c) = $dQ^p + dQ^{Eo}$)	+2.6	+2.5
Land use (Residual) dQ^R	+98.4	+97.5
Total change (observed) (dQ_{obs})	+101	

dQ^p and dQ^{Eo} are changes in runoff caused by precipitation and potential evapotranspiration respectively

4.3.3.1 *Runoff sensitivity to climate change*

Runoff sensitivity Equation (4.20) was developed for the watershed. The equation can be used to predict the expected relative change in runoff as a function of change in precipitation and potential evapotranspiration. The equation, for example, predicts that a 10% increase in rainfall would increase runoff by 20.7% while a 10% increase in potential evapotranspiration would reduce the runoff by 10.8%. Thus, it predicts that gain in runoff due to possible increase in rainfall would be minimized by possible increase in potential evapotranspiration.

4.3.3.2 *Expected future response of runoff due climate change*

Table 4.5 shows the calculated future estimates of potential evapotranspiration calculated using the IPCC projected change in temperature (Table 4.2) for the near-term (2016 - 2035) and medium-term (2046 - 2065) periods. The calculated values represent 4.2% and 5.3% increase in potential evapotranspiration for the near-term and medium-term periods respectively. The percentages were calculated based on the average potential evapotranspiration for the 1965 - 2007 period (Table 4.5). IPCC (2013a) projected an increase of 10% rainfall in the watershed region for both near-term and medium-term periods as shown in Table 4.5. The calculated percent change in PET and IPCC projected percent change in rainfall were applied in Equation (4.20) to predicted future response of runoff due to climate change, and the results are also shown in Table 4.5. The results indicate that the streamflow will increase by 16% and 15% for the near-term and medium-term periods due to climate change.

Table 4.5: Calculated PET and predicted change in runoff for near-term and medium term periods.

Period	**1965-2007**	**2016-2035**	**2046-2065**
PET (mm)	1556	1621	1638
Change in PET (%) (reference 1965-2007 period)		4.18	5.27
IPCC projected* change in precipitation (%)		10	10
Predicted change in runoff (%) - based on Equation (4.20)		16	15

**based on Representative Concentration Pathway (RCP4.5) - median (50% percentile) of the distribution of Coupled Model Inter-comparison Project Phase 5 (CMIP5) - (IPCC, 2013a)*

4.4 Discussion

4.4.1 Change in streamflow

It was concluded that land use change was the main driver of change in streamflow. The increasing trend in streamflow can be attributed to deforestation and conversion into agriculture in the Mau Forest and particularly the Eastern, South-western and Transmara blocks of the forest (Nkako et al., 2005). The forest blocks are at the headwaters of Nyangores River. Major deforestation and encroachment have been reported in this region. Mati et al. (2008) found that the forest cover in the Mara River basin was reduced by 32% between the years 1973 and 2000 while agriculture doubled over the same period. The Government of Kenya (GoK, 2009) estimated that in the larger Mau Forest complex block (Figure 4.1a), the closed canopy declined by 31% between 1973 and 2003 while the area under combined settlements and agriculture increased 5 times over the same period. Catchment water yield is likely to increase upon deforestation and conversion to agriculture although the extent depends on the scale, site and the level of degradation after conversion (Bruijnzeel, 2004; Calder, 2005). Other studies on paired catchment experiments have reported an increase in water yield after deforestation (Bosch and Hewlett, 1982; Mumeka, 1986; Sahin and Hall, 1996; Lal, 1997; Brown et al., 2005, 2013). Our results are also consistent with findings of a paired catchment experimental study by Recha et al. (2012). Their study catchment (Kapchorwa) under tropical rainforest of Nandi and Kakamega is also located within the Lake Victoria Basin in Western Kenya. They reported higher discharge for catchments that were deforested and converted to agriculture; the discharge also increased with time since deforestation.

The observed increase in discharge can be attributed to reduced evapotranspiration after deforestation (Bruijnzeel, 2004). This is because trees are generally known to have higher evapotranspiration than many other land uses, including agriculture (Calder, 2005). Comparatively, forests have higher interception 'losses', greater aerodynamic roughness and deeper roots – all which favour higher water use. The greater canopies of forests enable them to intercept and evaporate more rainfall while the extensive and deeper root network enhances their capacity to extract water from soil and groundwater storages (Bruijnzeel, 2004; Calder, 2005; FAO, 2006). In dry seasons, the tree roots, which are generally deeper than for most vegetation, act as 'pumps' that remove groundwater for transpiration (Bruijnzeel, 2004). Therefore, deforestation generally reduces vegetation water use in a watershed. The reduced 'pumping' of groundwater, particularly in dry seasons, make the water available for discharge inform of baseflow.

In Nyangores watershed, the observed increase in discharge was mainly contributed by increase in baseflow as shown in Figure 4.6 where the baseflow, separated using Web-based Hydrograph Analysis Tool (WHAT) recursive digital filter method (Eckhardt, 2012), followed a similar trend to the total discharge. This implies that at the annual level, the reduced evapotranspiration-showing as increased baseflow- is responsible for increased discharge.

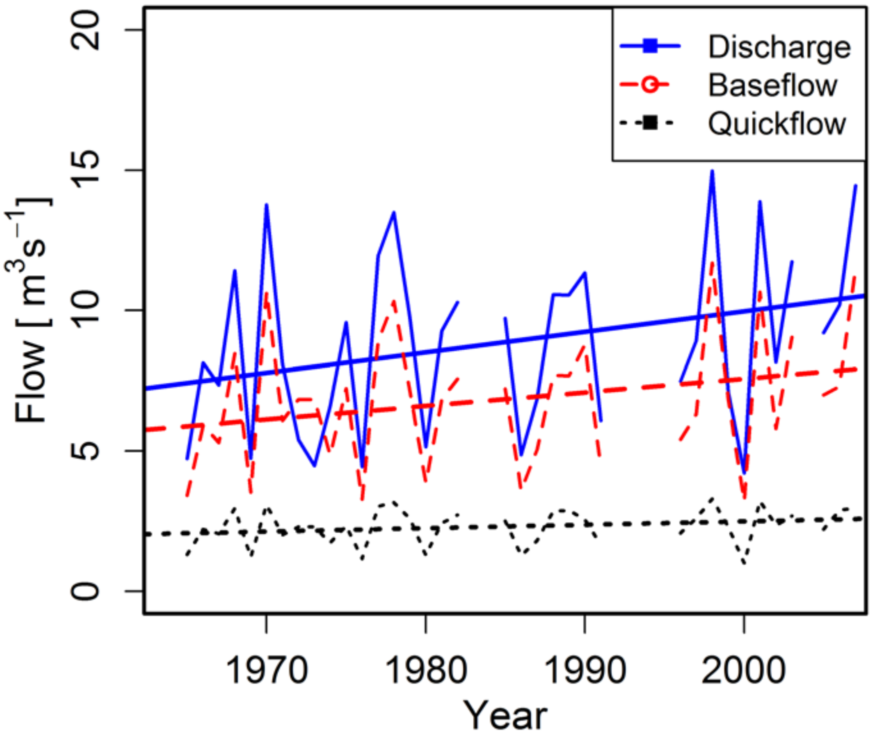

Figure 4.6: Temporal trends of annual baseflow, quickflow and total discharge of Nyangores River.

The breakpoint of the total annual discharge trend was found to be in 1977. Our findings are supported by Mati et al. (2005) who reported the increase in peak flows in Nyangores watershed starting in the same year, 1977. As shown in Figure 4.1, deforestation has been going on progressively in the watershed since the 1970's when there was massive land adjudication of the former communal *trust lands* in Kenya following the enactment of the Land Adjudication Act of 1968. The residents of Olenguruone *section* (Figure 4.1a) (formerly referred as Olenguruone settlement scheme) applied for land adjudication in 1976 (i.e., Land Adjudication Order, 1976 (Nakuru District)). The Olenguruone area, which is now under intensive cultivation, was formerly under dense natural forest and small pockets of montane open grassland (GoK, 1969; Muiru, 2012); grasslands, just like forest, have higher water infiltration capacities as compared with land under continuous cultivation (Gerla, 2007; Mao and Cherkauer, 2009; Heimann, 2009; Schilling et al., 2014; Everson, 2001). The colonial government that created the Olengurone settlement scheme in 1941 controlled the size and the location of land that the residents cultivated (Kanogo, 1987; Ochieng, 2009; Maxon and Ofcansky, 2014). After independence, in 1963, the restrictions were 'no more' and the locals abandoned the watershed conservation measures, put by colonial masters, which they deemed oppressive. At Olenguruone and the surrounding areas, increased acreages of land, including the hilly slopes, were put under cultivation which further increased with the land adjudication in the 1970s. The dense natural forest cover and the montane grassland in the area were cleared for cultivation and encroachment in the forest reserve started; all of which may have contributed to increase in discharge. Today, the area is under intensive subsistence agricultural cultivation and land ownership is a source of conflict among the ethnic communities living there. Indeed, Mati et al. (2008) found that the forest and grassland in the larger Mara River was basin reduced by 11% and 34% respectively between 1973 and 1986 while the area under open forest and cultivation increased by 73% and 96% respectively during the same period.

4.4.2 Attribution of changes in streamflow to changes in land use and climate variability

Climate variability was found to have only a minimal (2.5%) contribution to the observed change in discharge (Table 4.4). This can be attributed to the balance of the water input and atmospheric demand in the watershed. Both the water input (in form of precipitation) and the atmospheric water demand (in form of potential evapotranspiration) increased between the two periods. Thus, the total gain in discharge (24.2%) that would have been made by increased rainfall was reduced (by 21.6%) by the extra atmospheric water demand. On an annual basis, Nyangores can be classified as a water limited watershed (dryness index = 1.1). This implies that the available water (rainfall) does not fully satisfy the atmospheric water demand. The increase in rainfall between the two periods was also accompanied by a relatively higher increase in potential evapotranspiration (due to higher mean temperatures) which further raised the atmospheric water demand (i.e., further increasing the dryness index). Therefore, most of the extra rainfall was used up as evapotranspiration. Taking the effect of climate variability solely, actual evapotranspiration would have been expected to increase in the period after change point. However, as it can be seen in Table 4.3, the actual evapotranspiration decreased in the period after change point. The reduction in the estimated evapotranspiration between the two periods would then be attributed to change in catchment property (n). The change in catchment properties, occurring concurrently with climate variability, reduced the ‘would be’ gains in evapotranspiration in favour of increased runoff.

Change in catchment properties was found to be the main driver of the observed changes in runoff accounting for 97.5% of the change. Catchment properties that affect discharge are soil properties, vegetation and topography (Ward and Trimble, 2003; Yang et al., 2008; Price, 2011). Land use change affects these catchment properties and especially the former two in the case of deforestation. Therefore, the change in discharge caused by changes in catchment properties is equivalent to the changes caused by land use in this case. As highlighted in Section 4.4.1, the major land use changes in Mara River basin is deforestation and conversion to farmland which implies change of vegetation from natural tree vegetation to agricultural crops (mainly maize, beans and potatoes). Other than reduced water use, deforestation also exposes the land to degradation where soil properties are negatively affected eventually leading to reduction in water infiltration and increase in quick runoff. Soil-related factors that lead to decline in infiltration after deforestation include: compaction of top soil (increase bulk density), decrease in soil organic matter (reduce soil aggregation), decline in micro-faunal activity (reduces soil micro-pores), decrease in soil water holding capacities (Giertz et al., 2005; Celik, 2005; Recha et al., 2012).

The future watershed response of low flows to rainfall after deforestation depends on the balance between reduced evapotranspiration and the expected decrease in water infiltration due to degradation. If land degradation reaches a point where water infiltration is reduced to the extent that the quick flows exceeds the gain in baseflow, associated with reduced evapotranspiration after forest removal, then the dry season flows would decline. On the other hand, if the catchment properties do not change, i.e., no or minimal land degradation after forest removal and the original surface infiltration is maintained as before, then the effect of the reduced evapotranspiration may continue to be seen in high baseflow (Bruijnzeel, 2004; Brown et al., 2005). Thus, the observed increase in discharge and baseflow in Nyangores watershed may be short-lived depending on the future level of land degradation. There are already some signs of degradation in the cultivated areas of the watershed that were converted from the forests, as observed by runoff plot experiments by Defersha and Melesse (2012); they reported that cultivated lands in Nyangores watershed yielded higher sediment loads than other watersheds and land uses in the upper Mara River basin. It is also important to recognize that deforestation in the Mau Forest region has been progressive over time with more areas, illegally or legally, being carved out of the natural forest (Akotsi and Gachanja, 2004; Nkako et al., 2005; Akotsi et al., 2006; Mati et al., 2008; GoK, 2009; NEMA, 2013). Therefore, whereas the continued increase in discharge and baseflow may be due to accompanied decline in evapotranspiration, there may be some cultivated areas in the watershed facing high degradation, as observed by Defersha and Melesse (2012), whose response to rainfall may be quite opposite but their effect on baseflow being subdued. It is important therefore that efforts be made to arrest further deforestation and encroachment of the natural forests and more importantly to minimize degradation of the already deforested areas under cultivation.

The residual change in streamflow (dQ^R) may also contain, to a limited extent, change caused by intra-annual climate variability (Roderick and Farquhar, 2011). This is because the catchment property *n* encodes all factors that change the separation of P into E and Q under constant climate. Hence, other than change in land use discussed in this section, the changes in *n* over time may also be affected by factors such as changes in precipitation intensity or seasonal changes in precipitation and evapotranspiration (Roderick and Farquhar, 2011; Cuo et al., 2014; Zhang et al., 2015). For example, whereas an increase in dry season rainfall accompanied by an equal decrease in cold season rainfall may have no net change in annual rainfall (Onyutha et al., 2015), it may affect the separation of rainfall into runoff and evapotranspiration (Roderick and Farquhar, 2011). This is because the dry season generally has higher potential evapotranspiration than cold season and thus the change in evapotranspiration (occasioned by change in seasonal rainfall) for the two seasons may not completely balance at an annual scale. Seasonal variability in rainfall can be assessed by, for example, changes in quantiles (Ntegeka and Williems, 2008) or aggregation of rescaled series (Onyutha, 2015). However, since the change in streamflow caused by intra-annual variability is not separated from the residual change in streamflow dQ^R by the current version of Roderick and Farquhar, (2011) method used for this study, the seasonal changes in climate variability was not assessed; the qualitative description of its effect on *n* provided herein was considered sufficient and useful for further studies. We recommend use of more detailed hydrological models to compare the results obtained in this study.

In unregulated rivers like Nyangores, streamflow seasonality and persistence is more important measure of water availability than the total annual water yield (Döll and Schmied, 2012; Hoekstra et al., 2012; Bruijnzeel, 2004). Change in total water yield may also be accompanied by a change or shift in the seasonal streamflow (Brown et al., 2005; Zhang and Schilling, 2006). Although the study of streamflow seasonality is outside the scope of our paper, recent studies have reported that most downstream sections of the Mara River basin, which heavily rely on flow from the Nyangores River in dry seasons (McClain et al., 2014), are already facing water stress in dry months of the year (Dessu et al., 2014). Thus, further research on the effect of land use change on seasonal streamflow is highly recommended. Change in streamflow seasonality may be assessed by use of monthly/seasonal coefficient of variation (e.g. Zheng et al., 2007; Yang et al., 2009; Patil and Stieglitz, 2011) or non-uniformity coefficient (e.g. Li et al., 2014) and estimated by changes in seasonal/monthly flow duration indices (e.g. Li et al., 2014; Yang et al., 2009; Khaliq et al., 2008; Zheng et al., 2007).

4.4.3 Future change in runoff due to climate change

The runoff sensitivity Equation (4.20) calibrated for the watershed predicts that runoff is more sensitive to changes in precipitation than changes in potential evapotranspiration. Using the projected future climate change scenarios (Tables 4.2 and 4.5), the equation predicted that climate change would have a net increase in mean annual streamflow of 16% and 15% in the next 20 and 50 years, respectively (Table 4.5). The expected gains in discharge due to projected increase in rainfall would be reduced by the predicted increase in evaporative atmospheric water demand (Equation 4.20). The IPCC projected increase in temperature would essentially raise the atmospheric water demand (potential evapotranspiration), which would then buffer the 'expected' gain in runoff due to projected increase in rainfall. The predicted climate change-induced relative change in runoff for the next 50 years is slightly lower than for the next 20 years (Table 4.5). This is because whereas the IPCC projected an increase of mean monthly temperatures of about 0.5^0C between the two periods (Table 4.2), the rainfall increase remains constant at 10% (Table 4.5). Thus, the medium-term period would have a relatively higher PET and consequently less climate change-induced change in runoff as compared to the near-term period. The results indicate that direct climate change-induced change in streamflow is relatively moderate (i.e., 15% increase in 50 years). However, climate change may also have an impact in land use and human activities as people to adapt to the changes in climate. As already discussed, land use change has a major impact on both water yield and temporal pattern of streamflow and thus the effect may be greater than predicted.

We used the regional climate change projections based on the distribution of all the 42 models used in CMIP5. The purpose was to roughly show the sensitivity of runoff in the Nyangores based on general future projections. As already discussed in section 4.4.2, the runoff sensitivity model developed does not account for the intra-annual variability in climate which may also affect the predictions of runoff (Roderick and Farquhar, 2011). The predictions are thus approximate based on average values. We therefore did not select outputs from any specific GCM nor did we downscale the outputs of the 42 GCMs used in this study. The regional projections in temperature and rainfall used in this study, however, compare well with the values downscaled for the same study area by Dessu and Melesse (2013), and Akurut et al. (2014). Runoff predictions by this simple model are similar to that of the more detailed hydrological model implemented in SWAT by Mango et al. (2011). They reported that a future increase of about 10% in rainfall in the study area will have a modest increase in runoff due to increase in evapotranspiration, driven by accompanying rise in temperature. Unlike the complex hydrological models that demand much effort, data and time, the simple runoff sensitivity equation developed in this study can be easily used by water resources managers in the watershed.

It is also important to recognize the effect the uncertainties arising from the used IPCC future climate projections (Tables 4.2 and 4.5) would have on the results obtained in this study. The future temperature values used are based on projections of RCP4.5 scenario. RCP scenarios are based on predicted future forcing (RF) of the climate system by natural and anthropogenic forcing agents such as greenhouse gases, aerosols, solar forcing and land use change (IPCC, 2013b). The RCP4.5 scenario is based on estimated RF of 4.5 watts per square meter (W/m^2). However, the RF could fall outside this estimate depending on actual future emissions resulting from forcing agents. IPCC (2013b) gives different projections of temperature and rainfall for other estimates of RF (i.e., RCP2.5, RCP6.0 and RCP8.5) depending on the potential emissions from human activities and/or natural causes (e.g., volcano eruptions). To estimate the range of potential future change in streamflow, based on potential range of change in temperature and rainfall, future runoff prediction was carried out using the projections of the extreme climate change scenarios of RCP2.5 and RCP8.5 for medium-term period. For short-term period projection, the changes in temperature (i.e., 1^0C) and rainfall (i.e., 10%) are uniform across all the three RCP scenarios for the study area and therefore there would be no difference in the predicted change in streamflow (i.e., remains the same as for RCP4.5 (Table 4.5). As shown in Table 4.6 and compared with RCP4.5, lower future emissions (RCP2.5) will cause a slight increase in streamflow (to 16%) while higher emissions (RCP8.5) will reduce the potential gain of streamflow to 12.7 %. Thus, the predicted potential increase in runoff of 15% for the 2036 - 2065 period could fall anywhere in the range between 12.7% and 16.0% depending on the actual future emissions.

Table 4.6: Predicted change in runoff based on different IPCC emission projection scenarios

RCP Scenario ***(for the period 2046-2065)***	**RCP2.5**	**RCP4.5**	**RCP8.5**
PET (mm)	1624	1638	1671
Change in PET (%) ***(reference period: 1965 - 2007, PET =1556mm)***	4.4	5.3	7.4
IPCC projected change in precipitation (%)	10	10	10
Predicted change in runoff (%) - based on Equation (4.20)	16	15	12.7

4.5 Summary of results and conclusions

The relative impact of land use change and climate variability on streamflow at the Nyangores watershed in Kenya was investigated. The climate variability impact on streamflow was further partitioned into effects caused by changes in precipitation and those caused by changes in potential evapotranspiration. Future impact of climate change on streamflow was then projected. Quantification of the contributions of the observed change in streamflow of River Nyangores caused separately by land use change and climate variability is one of the main contributions of this study. Though there have been previous studies that have attributed change in hydrology of larger Mara River basin to land use change, information on how much of the observed change in historical streamflow record was caused by either land use change or climate variability has been lacking. Another unique contribution of this study is development of a simple runoff sensitivity equation that can easily be used by water resources managers in the watershed to estimate change in streamflow as a function of change in rainfall and potential evapotranspiration. Main findings and conclusions of the study are:

1. There is an increasing trend in the annual streamflow at the Nyangores watershed. Trend analysis using the Mann Kendall tests detected a significant increasing trend in annual streamflow. The breakpoint for the time series trend was found to be 1977 using the sequential Mann Kendall test.

2. Land use change is the main driver of the change in streamflow accounting for about 97.5% of the change. This can be attributed to the deforestation in the Mau Forest complex at the headwaters of the river. Forest removal and conversion to cropland agriculture caused the increase in streamflow due to reduced water use of crops as compared to forest. We recommend further study on the effect of land use change on seasonal flow regime of the river and its impact on the downstream water availability.

3. Climate variability contributed only a small percentage (2.5%) of the change of streamflow. There was an increase in both precipitation and potential evapotranspiration whose individual effect on streamflow change counters each other (increase in both water input and evaporative demand) resulting to a slight net change in runoff.

4. Streamflow change solely caused by climate change was predicted to increase by 16% and 15% for the next 20 and 50 years respectively. The effect of the predicted increase in rainfall on runoff would be offset, to some extent, by the expected increase in evaporative water demand due to projected increase in temperature. Judging from our findings of the last decades, land use change may still be the major driver of future change in streamflow and may overshadow the predicted impacts of climate change.

5. Deforestation is majorly responsible for change in Nyangores River hydrology. Thus, management measures that control further loss of natural forest and reduce degradation of farmland are required. Thus, the promotion of tree vegetation (e.g. as buffer strips or as integral part of agroforestry systems) may be helpful to mitigate the formation of surface runoff and associated soil erosion.

5 Chapter five: Modelling the impact of agroforestry on hydrology of Mara River Basin.

Publication (this chapter has been published in Hydrological Processes)

Mwangi HM, Julich S, Patil SD, McDonald MA, Feger KH. (2016). Modelling the impact of agroforestry on hydrology of Mara River Basin in East Africa. *Hydrological Processes* 30: 3139-3155.

Abstract:

Land–use change is one of the main drivers of watershed hydrology change. The effect of forestry related land–use changes (e.g. afforestation, deforestation, agroforestry) on water fluxes depends on climate, watershed characteristics and spatial scale. The Soil and Water Assessment Tool (SWAT) model was calibrated, validated and used to simulate the impact of agroforestry on the water balance in the Mara River Basin (MRB) in East Africa. Model performance was assessed by Nash-Sutcliffe Efficiency (NSE) and Kling-Gupta Efficiency (KGE). The NSE (and KGE) values for calibration and validation were: 0.77 (0.88) and 0.74 (0.85) for the Nyangores sub-watershed, and 0.78 (0.89) and 0.79 (0.63) for the entire MRB. It was found that agroforestry in the watershed would generally reduce surface runoff, mainly due to enhanced infiltration. However, it would also increase evapotranspiration and consequently reduce baseflow and overall water yield, which was attributed to increased water use by trees. Spatial scale was found to have a significant effect on water balance; the impact of agroforestry was higher at the smaller headwater catchment (Nyangores) than for the larger watershed (entire MRB). However, the rate of change in water yield with an increase in area under agroforestry was different for the two and could be attributed to the spatial variability of climate within the MRB. Our results suggest that direct extrapolation of the findings from a small sub-catchment to a larger watershed may not always be accurate. These findings could guide watershed managers on the level of trade-offs that might occur between reduced water yields and other benefits (e.g. soil erosion control, improved soil productivity) offered by agroforestry.

5.1 Introduction

Forests provide a number of ecosystem services, such as improving soil water infiltration conditions, soil erosion control and provision of wood–related products like timber and fuelwood (Calder, 2005; Ong et al., 2006). The fertility potential of soils under forests and the need to increase crop production makes forests a target for conversion to agricultural land through deforestation (Pope et al., 2015; Laurance et al., 2014). There is therefore high competition for land between forests and agricultural production in some regions of the world, particularly in the tropics (Laurance et al., 2014). In such situations, agroforestry is seen as a compromise between agricultural production and provision of forest/tree–related benefits (Garrity, 2012). In agroforestry systems, trees in different forms of arrangements are integrated into agricultural land (Nyaga et al., 2015; Nair, 1993). This kind of arrangement therefore ensures that the environmental services provided by the trees/forests are met to some extent, while at the same time agricultural land continues with its main role of crop production (Ong et al., 2006). Countries, particularly those whose economy mainly rely on agriculture, find agroforestry as a feasible means of increasing their forest cover and a way of controlling degradation of natural forests (Garrity, 2012). Indeed, this may be one of the best practical solutions of increasing tree cover in areas that have been deforested and settled by communities whose main source of livelihood is agricultural cultivation (Mbow et al., 2013). In such situations, complete afforestation may not be practical because people's livelihood is a priority.

With agroforestry, the question that arises is how much land can practically and sustainably be converted to tree cover. At the farm level, this trade-off is highly dependent on the extent of available land. However, at the watershed level, the trade-off and synergies between provisions of various ecosystem services is an important consideration (cf. Brauman et al., 2007).

For water resources managers, information on how and by how much agroforestry practices will affect water availability is pertinent. Determination of the thresholds of area of agroforestry (percent of tree cover) that would not compromise provision of watershed services is of paramount importance (Brown et al., 2005; Mwangi et al., 2015a). The question of how change in vegetation affects watershed hydrology is mainly centred around the impact on different components of catchment water balance. This is partly because different types of vegetation result in different levels of rainwater infiltration capacities. For example, forests are generally known to offer enhanced infiltration of rainwater compared to most other land-uses (Bruijnzeel, 2004). This is mainly brought about by a normally higher organic matter content and little anthropogenic disturbance in the forest soils as compared to, for example, cultivated lands. Therefore, the partitioning of rainwater into surface runoff and the water that infiltrates into the ground differs for landscapes with different types of vegetation - even in cases where soil type is similar. Plant water use (transpiration) also differs with vegetation type (Jian et al., 2015; Julich et al., 2015). Some vegetation, especially trees, often consumes more water than other vegetation (Albaugh et al., 2013; Julich et al., 2015). The rooting depth of vegetation also determines the depth to which plants of a particular type can draw water especially in the dry seasons when the top soil is dry (Thomas et al., 2012; David et al., 2013). Deep-rooted vegetation is able to extract groundwater from deeper aquifers particularly when the water table is low compared with shallow rooted vegetation (Pinto et al., 2014; Nosetto et al., 2012). Consequently, the extent of groundwater removal by vegetation of different types influences the amount of groundwater released to the streams as baseflow (Salemi et al., 2012). Water extraction by deep-rooted vegetation reduces groundwater storage and decreases the amount released to streams (Fan et al., 2014). It is therefore obvious that introduction of trees into crop lands (agroforestry) would cause changes in a watershed's water balance (Palleiro et al., 2013; Ong et al., 2006). The direction and magnitude of the change in different water balance components may differ with the watershed characteristics (e.g. soil, topography), climate, agroforestry tree species and more importantly, the proportion of the watershed under tree cover (Brown et al., 2013; Julich et al., 2015).

Field studies on the hydrological impacts of agroforestry (e.g. Zhao et al., 2011; Ghazavi et al., 2008; Muthuri et al., 2004; Radersma and Ong, 2004) have demonstrated the need to include (or improve) tree water uptake (transpiration) and canopy interception in watershed modelling. Ghazavi et al. (2008), for example, observed decreasing water table levels near hedgerows during the growing season (spring and summer) in Brittany, France which they attributed to high transpiration by hedgerow trees. A modelling study (using Hydrus-2D model) for the same site by Thomas et al. (2012) showed that transpiration is a substantial component of water balance representing 40% of total water output. Similar conclusions were drawn by Muthuri et al. (2004) who modelled water use by agroforestry systems in Nyeri County, Kenya, using the WaNuLCAS (Water, Nutrient and Light Capture in Agroforestry Systems) model.

In this study, we use the SWAT (Soil and Water Assessment Tool) model (version 2012) to assess the impact of agroforestry on hydrology of the Mara River Basin. Although SWAT has been extensively used for land–use change studies, its use for agroforestry simulation studies is not well documented. The Mara River basin is located in East Africa and has undergone significant land–use changes over the last 50 years, particularly deforestation and conversion to agriculture in the headwaters (Mati et al., 2008). Intensive cultivation is currently predominant in the formerly forested areas and the Government of Kenya (GoK) is keen on restoring forest cover in as much area as possible (GoK, 2009; NEMA, 2013). Considering that the basin is now densely settled by communities whose livelihood depend on agricultural cultivation (Kanogo, 1987), one of the feasible solutions to increase the tree cover in the upper Mara basin is through agroforestry (Atela et al., 2012; KFS, 2009). Because Mara is a trans-boundary river basin between Kenya and Tanzania, the upstream watershed activities, including land–use changes, are of interest not only to Kenya but also to Tanzania (Gereta et al., 2009). A thriving tourism industry in the shared Maasai Mara (Kenya) and Serengeti (Tanzania) game reserves ecosystem is also heavily dependent on the water resources provided by the Mara River (Gereta et al., 2002). For this reason, prediction of the effect of agroforestry on the water balance of the Mara River basin is paramount for sustainable water resources management.

5.2 Methods

5.2.1 Study area

The Mara River Basin (henceforth referred to as MRB) covers a total area of about 13,750 km^2, which is shared between Kenya (65%) and Tanzania (35%) (Figure 5.1). The two main headwater tributaries of the Mara River (Nyangores and Amala) originate from the Mau Forest and join on the Kenyan side of the border with Tanzania to form the main Mara River which drains into Lake Victoria on the Tanzanian side of the border. There are three main gauging stations within MRB: 1LA03 at Bomet (for Nyangores sub-watershed), 1LA02 at Mulot (for Amala sub-watershed), and Mara mines in Tanzania (for the larger MRB) (Figure 5.1). The drainage areas at the three outlets are: 692 km^2, 695 km^2 and 10,550 km^2 for Nyangores, Amala and Mara watersheds, respectively. The elevation of MRB ranges from about 3,000 m asl at the source in the Mau Forest complex to about 1100 m asl as the river drains into Lake Victoria. The basin experiences a bimodal rainfall pattern that varies with altitude. There are two main rainfall seasons i.e. from March to June (long rains) and from September to November (short rains). The mean annual rainfall ranges from about 1800 mm in the forested headwaters to about 600 mm in the downstream sections of the basin.

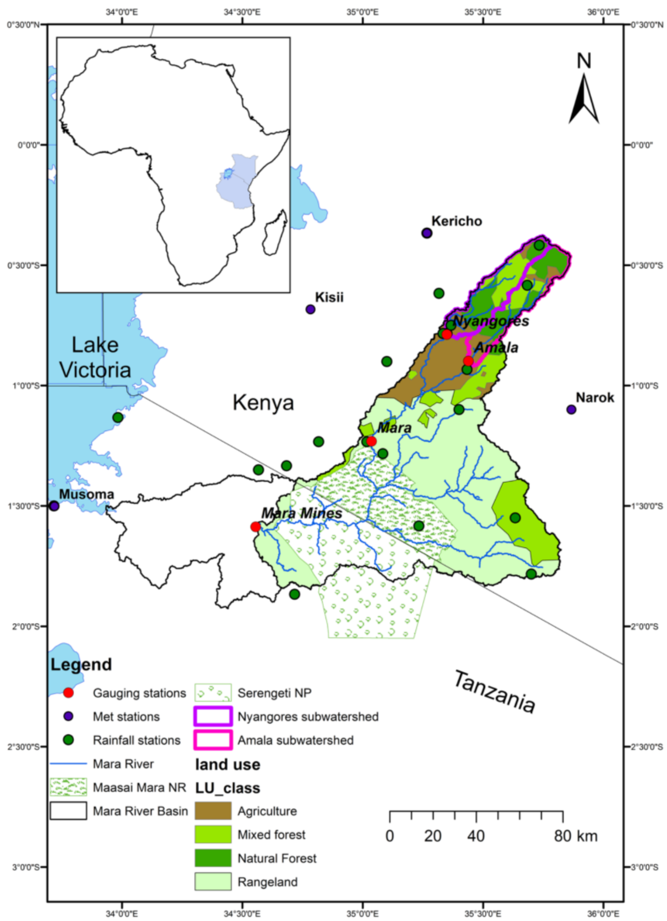

Figure 5.1: Mara River Basin.

Forests and agriculture are the main land-uses in the upstream region of the MRB (Figure 5.1). Pastoralism and wildlife conservation (in Maasai Mara and Serengeti National Reserves) dominate the middle sections of the basin (Figure 5.1). The areas adjacent to the game reserves are mainly used for livestock grazing and also as wildlife dispersal areas through some arrangements (e.g. conservancies) with the local pastoral communities (Osano et al., 2013; Homewood et al., 2012; Ogutu et al., 2009; Thompson and Homewood, 2002). The downstream region of the MRB in Tanzania is mainly dominated by subsistence agriculture and gold mining. The main soil types (World Reference Base classification) are: Planosols (30%), Phaeozems

(26%), Andosols (12%) Vertisols (10%), and Cambisols (9%). Other soils (13%) are: Leptosols, Luvisols, Nitisols, Greyzems, and Regosols.

5.2.2 SWAT Model

SWAT is a physically based, semi-distributed, meso-scale-watershed model (Arnold et al., 1998) widely used for prediction of the impact of land management on water, sediment and agricultural chemical yields (Neitsch et al., 2011; Gassman et al., 2007). The main inputs of the model are: Digital Elevation Model (DEM), land-use, soil, and climate data. SWAT first sub-divides a watershed into sub-watersheds which are further partitioned into smaller Hydrologic Response Units (HRU). Each HRU in a sub-basin has unique land-use, soil type, and slope class combination. Simulation of agroforestry scenarios in this study was based on HRU.

5.2.3 Model parameterization: SWAT input data

Climatic data was obtained from the Kenya Meteorological Department and the Tanzania Meteorological Agency. Daily rainfall data from 20 stations within and in close vicinity of the watershed (Figure 5.1) was used. For the selected period of model calibration and validation, the rainfall data was nearly 100% complete for more than half of the stations. Daily data sets of the other climatic variables i.e. maximum and minimum temperature, humidity, radiation and wind speed were obtained for Narok, Kisii, Kericho, and Musoma meteorological stations (Figure 5.1). For short gaps, missing data for a particular day was filled by arithmetic mean observed for the day in the neighbouring stations, whereas longer gaps (more than 10 days) were filled using the weather generator model, WXGEN, incorporated in SWAT (Neitsch et al., 2011) that relies on monthly mean values.

Shuttle Radar Topography Mission, 90-m, DEM was used for watershed delineation in SWAT. Soil data (scale of 1:1 million) was obtained from Kenya Soil Survey and Soil and Terrain Database (SOTER) of the International Soil Reference and Information Centre (ISRIC) (Batjes, 2002). Some soil parameters that were not available from the databases, e.g. saturated hydraulic conductivity, were estimated using pedotransfer functions (Nemes et al., 2005). A Land–use map of 1983 (GoK, 1983) was used for model setup (Figure 5.1). The map was compared with Landsat satellite images and land–use maps by Mati et al. (2008) for the same period. The land–use map was deemed appropriate for periods used for calibration and validation of the SWAT model. The proportions of the land-uses are: 10%, 21% and 69% for agriculture, forests, and rangeland, respectively (Figure 5.1).

Discharge data (for Nyangores, Amala and Mara (at Mara mines) Rivers) was obtained from Water Resources Management Authority (WRMA) in Kenya and the Ministry of Water in Tanzania. Nyangores data was 100% complete for both periods of calibration and validation. Mara and Amala data had gaps which were left unfilled.

5.2.4 Model parameterization: Plant growth

Although SWAT has been widely used for land–use study in tropical watersheds, its plant growth module is better suited for temperate regions. As such, it has some shortcomings in modelling the growth of trees and perennial crops in tropical regions (Wagner et al., 2011). This is because, unlike in the temperate regions, plants in the tropics do not have a dormant period and there is no seasonal shedding and sprouting of leaves for perennials. For this reason, the robustness of the model and the accuracy of the output based on default plant growth module parameters in the tropics or absence of information on its parameterization altogether, has been criticized (van Griensven et al., 2012; Strauch and Volk, 2013). Plant growth in SWAT is based on the heat unit theory which postulates that plants require a specific amount of heat to bring them to maturity (Neitsch et al., 2011). Thus, SWAT accumulates heat units from planting and maturity is reached when the plant-specific total heat units (PHU) are attained. A heat unit is equivalent to each degree of daily mean temperature above a base temperature (plant-specific temperature below which there is no growth).Thus, PHU is the summation of all heat units from planting to maturity. However, for perennials and trees, PHU are the accumulated heat units between budding and leaf senescence (Strauch and Volk, 2013; Neitsch et al., 2011). At the end of the growth cycle, plant stops transpiring and uptake of water and nutrients (in SWAT) (Neitsch et al., 2011). The repeat of the growth cycle for perennials and trees is triggered either by dormancy (in-turn triggered by latitude-dependent shortening of day length) or use of ‘kill’ operation (Strauch and Volk, 2013; Neitsch et al., 2011). When the growth cycle is restarted, the accumulated heat units drop to zero and the leaf area index (LAI) is set to minimum. LAI partly controls water uptake by plants (transpiration) in SWAT. In SWAT, actual transpiration is calculated from potential evapotranspiration (PET) by Equation 5.1 when PET is simulated using the Priestley and Taylor (Priestley and Taylor, 1972), or the Hargreaves methods (Hargreaves and Allen, 2003). Thus, when LAI drops to minimum values, transpiration reduces accordingly.

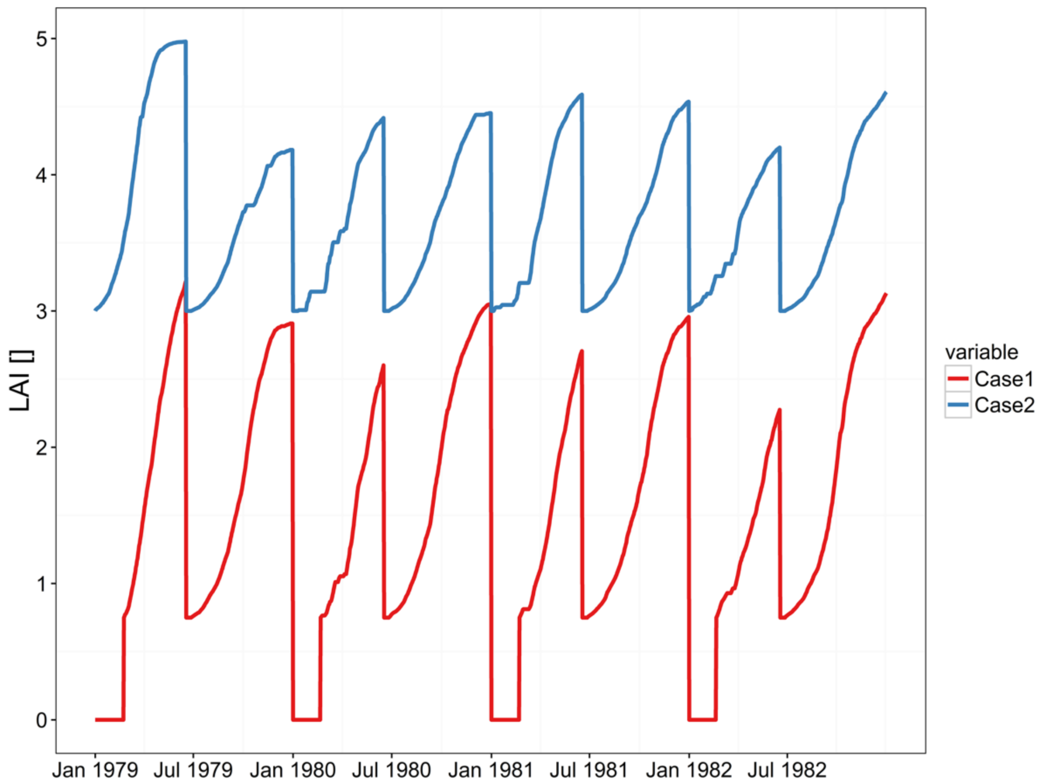

Figure 5.2: Leaf Area Index simulated using a) case 1: the default setting in SWAT (Minimum LAI = 0.75; 'start growing season' PHU fraction = 0.15) and b) case 2: adjusted values (Minimum LAI = 3.00; 'start growing season' PHU fraction = 0.001). PHU = 3500

$$E_t = \frac{E_0'.LAI}{3} \qquad 0 \le LAI \le 3$$

$$E_t = E_0' \qquad LAI > 3 \tag{5.1}$$

Where E_t is the maximum transpiration on a given day (mm), E'_0 is the potential evapotranspiration adjusted for evaporation of free water in the canopy (mm), and *LAI* is the leaf area index.

To adapt plant growth for our study site, the 'kill' operation was used to restart the growth cycle for trees and perennials. The minimum LAI for trees was increased from the default 0.75 (which is based on tree physiology in temperate regions) to 3.0 which is typical for the region (Broadhead et al., 2003; Muthuri et al., 2005). This ensured tree water use does not go unrealistically low in this tropical watershed. In SWAT, growth is initiated after a certain PHU fraction is attained. For this study, this fraction was reduced from the default of 0.15 to a small value of 0.001 to ensure that growth starts immediately after the growth cycle begins and allow for continuous transpiration. Simulated forest LAI using SWAT default values and the adjusted values (Figure 5.2) clearly indicate that the default values do not represent the growth that is typical in the tropics and therefore underestimates transpiration (Equation 5.1). The Priestley and Taylor (1972) method was used for calculation of transpiration in this study. Considering data availability and quality in the study site, this method was preferred because it uses less climatic data, unlike the widely used Penman-Monteith method (Allen et al., 1998) which is data intensive. The Priestley and Taylor (1972) method has been found to give better results than many other methods (e.g. Lu et al., 2005; Ding et al., 2013; Juston et al., 2014) particularly in areas with data availability or quality challenges.

The adjusted values (case 2 in Figure 5.2) were considered to provide a better representation of the leafing phenology and tree water use reported from field studies conducted in the region (Broadhead et al., 2003; Muthuri et al., 2004; Radersma et al., 2006; Ong et al., 2007). The seasonal variation in LAI also matched the bimodal pattern of rainfall in the watershed with minimum LAI coinciding with the dry seasons of January-February and July-August when growth is limited by moisture availability. July and August are also the coldest months of the year further limiting plant growth. Maximum LAI, and by extension high evapotranspiration, coincided with the long and short rains which was well simulated by the two cycles of leaf flush of trees observed in this region (Muthuri et al., 2004; Broadhead et al., 2003).

5.2.5 Calibration

Streamflow data for Nyangores tributary and the main Mara River at Mara mines were used for calibration and validation while that of Amala tributary was used for validation only. Nine years of daily streamflow data was used: four years (1979-1982) for calibration and five years (1974-1978) for validation. A two-year ‘warm up’ period was allowed for both calibration and validation. Calibration and validation periods also included dry and wet years and therefore low and high flows were well represented. The selection of this period (1974-1982) was guided by consideration of the completeness and degree of confidence of both meteorological and streamflow data at the three gauging stations. The main gauging station at Mara Mines has no recent streamflow data sufficient for model calibration; the data after 1990 is largely missing (McClain et al., 2014; Melesse et al., 2008). Of the two upper tributaries, the available streamflow data for Amala is of lower quality compared with that of Nyangores (Dessu and Melesse, 2012). It has many and long gaps; and we further established that the data had higher uncertainty for the period after 1980, arising from faulty rating equations. The land-use map used (for 1983) was considered appropriate to represent the land conditions during the calibration and validation periods. During the model setup, the MRB (up to Mara mines gauging station) (Figure 5.1) was subdivided into 92 sub-basins. The spatial variability of the watershed conditions in the MRB was taken into account during the model calibration which was done in two stages: first for the sub-basins in the upper Mara and then for the larger basin without changing the calibrated parameter values obtained for the upper Mara sub-basins.

For the upper Mara, calibration was done with the main outlet at Nyangores River at Bomet and the corresponding measured streamflow data for the station was used. After calibration and validation of the Nyangores sub-watershed, the optimized parameter set was then transferred to the neighbouring Amala sub-watershed. The two sub-watersheds are similar in topography, size, land–use, soils and climate. Due to its low quality, the streamflow data for Amala was only used for validation, and the validation period was prior to 1980 when the data quality was better. Considering the low quality of observed streamflow data for Amala and taking advantage of its topographical similarity with Nyangores, we sought to investigate how well the model parameters calibrated for Nyangores would perform when transferred to Amala.

Calibration parameters (Table 5.1) were identified by sensitivity analysis and Latin hypercube sampling was used to select sets of parameter values for automatic calibration using Particle Swarm Optimization (PSO) algorithm (Kennedy and Eberhart, 1995; Eberhart and Shi, 2001). The principle of PSO is based on the social behaviour of a population of particles (i.e. swarms such as flocking birds) moving towards the most promising area of the search space (e.g. location of food) (Reddy and Kumar, 2007). PSO is initialized using a group of random particles (e.g. through Latin hypercube sampling) with each particle representing a possible solution. Each potential solution is also assigned a randomized velocity which directs the ‘flying’ of the particles (Eberhart and Shi, 2001; Reddy and Kumar, 2007). The potential solutions are then “flown” through the problem space (Eberhart and Shi, 2001; Shi and Eberhart, 1998). At the end of each iteration, the position and velocity of a particle (i.e. parameter set) are updated. The position represents the current value within the search space and velocity represents the direction and the speed the search is moving in (i.e. rate of change in the dimensional space). The positions of the particles are changed (updated) within the search space based on the social tendency of the individuals (particles) to emulate the success of other individuals (Reddy and Kumar, 2007). All the particles have fitness values which are evaluated by the objective function to be optimized. Each particle keeps track of its coordinates in the solution space which are associated with the best solution (fitness) it has achieved so far i.e. the 'pbest'. PSO also tracks the best solution achieved at any point by any particle in the population (swarm) which is referred to as global best solution ('gbest') (Poli et al., 2007; Eberhart and Shi, 2001). Each PSO iteration aims to move each particle, by changing its velocity, closer to its personal best ('pbest') position and the global best position. After several iterations, one good solution (optimized) is produced when the particles converge towards the global optima.

Table 5.1: Calibrated SWAT model parameters

Parameter		**Calibrated parameter values**			**Parameter range used for calibration**	**Description**	
		common	Upper Mara	MRB			
Surlag		3.74			0 - 4	Surface runoff lag coefficient	
AWC*		0.14			-0.20 – 0.20	Available water capacity of soil	
CN	FRSE	35			35 - 40	Initial Soil conservation service (SCS) runoff curve number for moisture condition II	Forest 'evergreen'
	FRST	36			35 - 40		Mixed forest
	AGRR	60			60 -75		Agriculture
	SWHT	71			60 - 75		wheat
	RNGE	36			35 - 45		rangelands
CH_N			0.12	0.09	0.01-0.3	Manning's "n" value for the main channel	
CH_K			3.23	2.98	0-10	Effective hydraulic conductivity in main channel alluvium (mm/hr)	
ALPHA_BF			0.75	0.98	0.6 - 0.99	Base flow alpha factor (l/days)	
GW_delay			31.0	4.91	0 - 31	Ground water delay time (days)	
GW_Revap			0.14	0.10	0.02 - 0.15	Groundwater "revap" coefficient	
GWQMN			200	1869	150 - 2000	Threshold depth of water in the shallow aquifer required for return flow to occur (mmH_2O)	
Rchrg_dp			0.25	0.10	0.02 - 0.25	Deep aquifer percolation fraction	

*percent of the parameterized soil awc for layer of each soil

5.2.6 Evaluation of model performance

Goodness-of-fit (fit-to-observation) was used as the main criterion for evaluation of model performance (Moriasi et al., 2007; van Griensven et al., 2012). In addition, we also evaluated the catchment water balance in order to ensure the various components (e.g. runoff, evapotranspiration and groundwater contribution to streamflow) were within reasonable ranges typical of the study area. The aim was to ensure a realistic representation of hydrological processes and watershed conditions of the MRB (fit-to-reality). We also aimed to ensure that the calibrated model was fit for the intended purpose of land-use change simulation (fit-to-purpose) (van Griensven et al., 2012). We focused on selection of realistic ranges of the model input parameter values in order to reduce uncertainty in the model outputs (Arnold et al., 2012). Selection of realistic ranges of SWAT input parameters prior to calibration has been shown to reduce model prediction uncertainties (Zhenyao et al., 2013; Benaman and Shoemaker, 2004). We particularly paid special attention to model parameters that govern the water 'loss' from the system e.g. *CH_K, GWQMN, GW_Revap* and *Rchrg_dp* (Table 5.1) (Neitsch et al., 2011). Wrong selection of these parameter values may lead to unrealistic water balance even when there is a good fit between observed and simulated streamflows. For example, high values of *Rchrg_dp* may lead to high deep percolation losses which may be compensated by unrealistically low levels of evapotranspiration, even in cases where streamflow is within the ranges of observed values. Qualitative and quantitative guidelines on the appropriate ranges of these parameters for the study region are given by van Griensven et al. (2012).

Knowledge of the watershed is important in hydrologic modelling as there is no automatic procedure of parameterization and calibration which can substitute for actual physical knowledge (Arnold et al., 2012). Zhenyao et al. (2013) studied the impact of parameter distribution uncertainty on hydrological modelling using SWAT and recommends use of any available knowledge of the watershed to aid selection of realistic parameter ranges to reduce prediction uncertainties. Besides the guidelines by van Griensven et al. (2012), we used our watershed knowledge as well as knowledge from our past experience of SWAT application in the region (e.g. Gathenya et al., 2011, Mwangi et al., 2012a, Mwangi et al., 2015c) and literature of SWAT application in the area (e.g. Dessu and Melesse, 2012; Mango et al., 2011; Githui et al., 2009; Baker and Miller, 2013) to select reasonable SWAT input parameter ranges (Table 5.1). In addition, preliminary model runs were used to guide the selection of the parameter ranges that represent reasonable water balance conditions of the watershed. Typical ranges of water balance components e.g. surface runoff, baseflow and evapotranspiration were also assessed based on the knowledge of the watershed as well as published literature in the region (e.g. Dagg and Blackie, 1965, 1970; Krhoda, 1988; Water Resources and Energy Management, 2008; Mati et al., 2008; Mutiga et al., 2010; Recha et al., 2012; Dessu and Melesse, 2012; Baker and Miller, 2013; Mwangi et al., 2016b). Dagg and Blackie (1965, 1970) reported that 'deep percolation loss' was minimal for their experimental study site in Mau forest. This information, for example, guided us in setting up the upper limit for the parameter *Rchrg_dp* and results of 'deep percolation loss' from preliminary model runs helped in adjusting the parameter value range. In another example, our previous study in the watershed (Mwangi et al., 2016b) showed that baseflow constitutes a large percentage (ca. 80%) of the streamflow of Nyangores sub-watershed. We used this information to evaluate the water balance components in our preliminary model runs and adjust relevant ranges of the relevant input parameters e.g. *GWQMN* (Table 5.1). Further model parameterization, particularly regarding the adaptation of the plant growth module for the watershed, ensured that the calibrated model was fit for the purpose (i.e. land–use simulation).

Statistical fitting of the simulated and observed streamflow was then used for model performance evaluation during automatic calibration. Nash-Sutcliffe Efficiency (NSE) (Nash and Sutcliffe, 1970) was used as the objective function in the PSO algorithm. NSE is a normalized statistic ranging from - ∞ to 1 and is calculated as follows:

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}\{q_{obs}(t) - q_{sim}(t)\}^2}{\sum_{i=1}^{n}\{q_{obs}(t) - q_{meanObs}\}^2}\right] \quad (5.2)$$

where $q_{obs}(t)$ is the observed discharge at time step t, $q_{sim}(t)$ is the simulated discharge at time step t, $q_{meanObs}$ is the mean of the observed discharge over the simulated period, and n is the total number of observations.

One limitation of NSE is that it underestimates peak flows and overestimates low flows (Gupta et al., 2009). In light of this, a second objective function, Kling–Gupta Efficiency (KGE) (Gupta et al., 2009), was used for evaluation of model performance to overcome the weakness of NSE. The KGE statistic is based on the decomposition of model error into three distinct components which measure the linear correlation, the bias and the variability of flow respectively (Gupta et al., 2009; Kling et al., 2012). The latter two components relate to the ability of the model to reproduce the distribution of flow as summarized by first and second moments (i.e. mean and standard deviation) while the former relate to the ability to reproduce the timing and shape of the hydrograph. KGE is calculated as follows:

$$KGE = 1 - \sqrt{(r-1)^2 + (\alpha-1)^2 + (\beta-1)^2} \tag{5.3}$$

$$\alpha = \frac{\sigma_{sim}}{\sigma_{obs}} \tag{5.4}$$

$$\beta = \frac{\mu_{sim}}{\mu_{obs}} \tag{5.5}$$

where *r* is the correlation coefficient between simulated and observed streamflow, α is the variability ratio, β is the bias ratio, σ and μ are the standard deviation and the mean of the streamflow respectively, and indices *sim* and *obs* represent simulated and observed values of streamflow respectively.

5.2.7 Simulation of agroforestry

After calibration and validation of the SWAT model, agroforestry land–use scenarios were simulated. The structure of the SWAT model allows only one plant or crop type per HRU. The most typical systems of agroforestry in the watershed are: 1) intercropping sparsely distributed trees with different crops, 2) trees along the hedges and borders, and 3) woodlots (Nyaga et al., 2015; Lagerlöf et al., 2014). The first two agroforestry systems posed a challenge to be explicitly implemented in SWAT due to the model structure. Thus, agroforestry was implemented as woodlots at the HRU level. Woodlots have recently become popular in Kenya due to high demand for wood products (Nyaga et al., 2015). The woodlots were considered to offer, at the watershed level, a general spatial representation of the practical agroforestry system. Additionally, the hydrological impact (i.e. water use and infiltration characteristics) of agroforestry at the farm level was, to a larger extent, captured at the sub-basin level.

The agroforestry scenarios were simulated on land currently under cultivated agriculture implemented on a SWAT project based on 2014 land-use (Lariu, 2015). The selection of agroforestry scenarios was based on tree cover increment in the MRB upstream of Mara gauging station (Figure 5.1). To increase the area under agroforestry (tree cover), the number of HRUs with trees (forest) were increased by conversion of some HRUs previously under agriculture to woodlots (i.e. pure tree stand with properties of a forest). To implement this in SWAT, we considered slope as a practical criteria which additionally provided another advantage of maintaining the same HRU configurations across all the scenarios. We therefore selected four slope classes i.e. 0-10%, 10%-15%, 15%-20% and over 20% when creating the HRUs and which were later used as the basis for implementing the agroforestry scenarios.

All the scenarios were assessed relative to the base scenario that represents the current land-use/cover (for year 2014) in the basin (Figure 5.3a). For clarity, Figure 5.3 only shows maps of land–use and agroforestry scenarios for the Nyangores sub-watershed. The first scenario was implemented by changing all the HRUs under cultivated agriculture, that fall within the slopes above 20%, to woodlots (Table 5.2). Similarly, the second scenario was simulated by converting the HRUs in the slope category of 15 - 20% which were under cultivated agriculture to woodlots. So, in total for this scenario, all the agricultural HRUs in slopes >15% were simulated as woodlots. The same was done for the slope class of 10 - 15% for the third scenario. Other than change in vegetation, infiltration properties of the target HRUs were also adjusted from that of agriculture to that of forest. This was accomplished in SWAT by change in curve number (Table 5.1). The curve number is a parameter of the United States Soil Conservation Service (SCS) empirical equation (SCS, 1972) used for estimation of surface runoff. It is a function of soil permeability, land–use, and antecedent soil water conditions. Agricultural HRUs in the slope class of 0-10% were not converted to woodlots as that would have simulated complete afforestation of the upper Mara watershed which was not the objective of this study and is neither practical in this area where smallholder agriculture is the main source of livelihood (Atela et al., 2012).

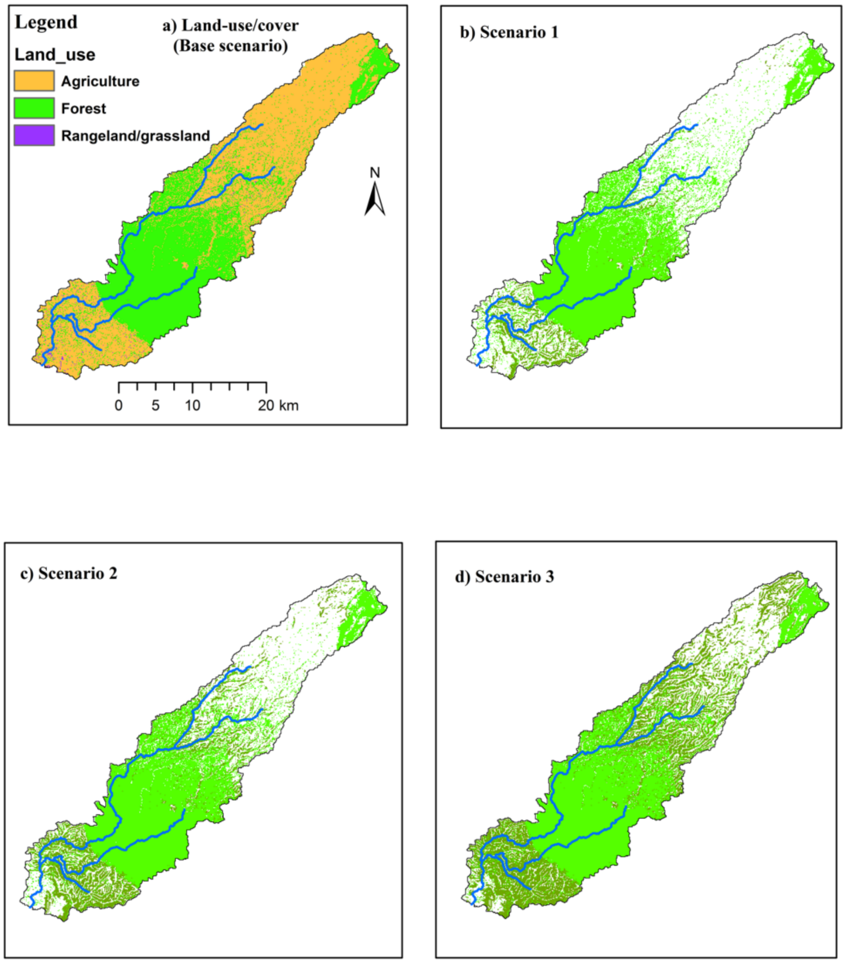

Figure 5.3: Land–use and agroforestry scenario maps (for Nyangores sub-watershed only): (a) Land–use/cover map (2014); also represents the base scenario. (b, c, and d) Agroforestry scenarios 1, 2, and 3 respectively (showing the forest cover in the base scenario (light green) and additional areas simulated with woodlot agroforestry (dark green).

Table 5.2: size of the watershed converted to forest under the three agroforestry scenarios

	Mara River Basin (area = 10,550 km^2)		
Scenario	Lower slope threshold (%)	area (ha)	% of watershed area
S1	20	18,559	1.8
S2	15	34,321	3.3
S3	10	63,810	6.0
	Nyangores sub-watershed (area = 692 km^2)		
Scenario	Lower slope threshold (%)	area (ha)	% of watershed area
S1	20	4,420	6.4
S2	15	9,965	14.4
S3	10	19,380	27.9

5.3 Results and Discussion

5.3.1 Calibration and validation

The monthly NSE (and KGE in parenthesis) values obtained for calibration are: 0.77 (0.88) and 0.78 (0.89) for Rivers Nyangores and Mara respectively (Table 5.3). The validation NSE (and KGE) are: 0.74 (0.85), 0.75 (0.68) and 0.79 (0.63) for Rivers Nyangores, Amala and Mara respectively. These values indicate that the SWAT model performance for this study was better compared to other previous studies in the watershed (Mango et al., 2011; Dessu and Melesse, 2012). This is probably due to better representation of tree growth (particularly with regard to water use) for tropical conditions. Unrealistic representation of forest transpiration in the tropics has been cited as one of the possible causes of marginal effect of forest-related land–use change on water balance in some of the previous simulation studies (e.g. Mango et al., 2011; Githui et al., 2009) conducted in the region (van Griensven et al., 2012). Separate calibration of the headwater sub-watersheds before including the rest of the basin may have also improved the model performance in this spatially-variable watershed. The daily hydrographs (Figure 5.4) show that the model, to a higher level of degree, reproduced the observed streamflow at all the gauging stations and thus well represented the rainfall runoff processes of the basin. This is confirmed by percent bias (PBIAS) which measures the average tendency of the simulated streamflow being larger or smaller than observed streamflow (Gupta et al., 1999). The PBIAS values for calibration (and validation in parenthesis) are: 1.3% (-8.9%), -0.12% (-34%) and (3.9%) for Rivers Nyangores, Mara and Amala respectively. All the values except for validation at Mara mines are within the ± 25% range proposed by Moriasi et al. (2007) for satisfactory calibration. The slight overestimation of average flow for Mara mines could be caused by slightly higher simulated peak flow which is also visible in the other hydrographs (Figure 5.4). This may be caused by the uncertainty in the observed streamflow data arising from inability to accurately measure high flows in the manual river gauging stations or from rating equations when converting gauge heights of high flows to discharge (cf. Juston et al., 2014). The model performance at Amala sub-watershed (Table 5.3; Figure 5.4c) implies watershed characteristics that are similar to Nyangores and that the streamflow of Nyangores River can be used to infer the hydrology of Amala sub-watershed (Klemeš, 1986). The validated model could also be useful in correcting streamflow records for Amala River for the period after 1980.

Table 5.3: Daily and monthly Nash-Sutcliffe efficiencies (NSE) and Klingupta efficiencies (KGE)

	Calibration			
	Daily		Monthly	
Gauging station	NSE	KGE	NSE	KGE
Nyangores	0.65	0.81	0.77	0.88
Maramines	0.46	0.72	0.78	0.89
	Validation			
Nyangores	0.63	0.80	0.74	0.85
Maramines	0.56	0.52	0.79	0.63
Amala	0.67	0.67	0.75	0.68

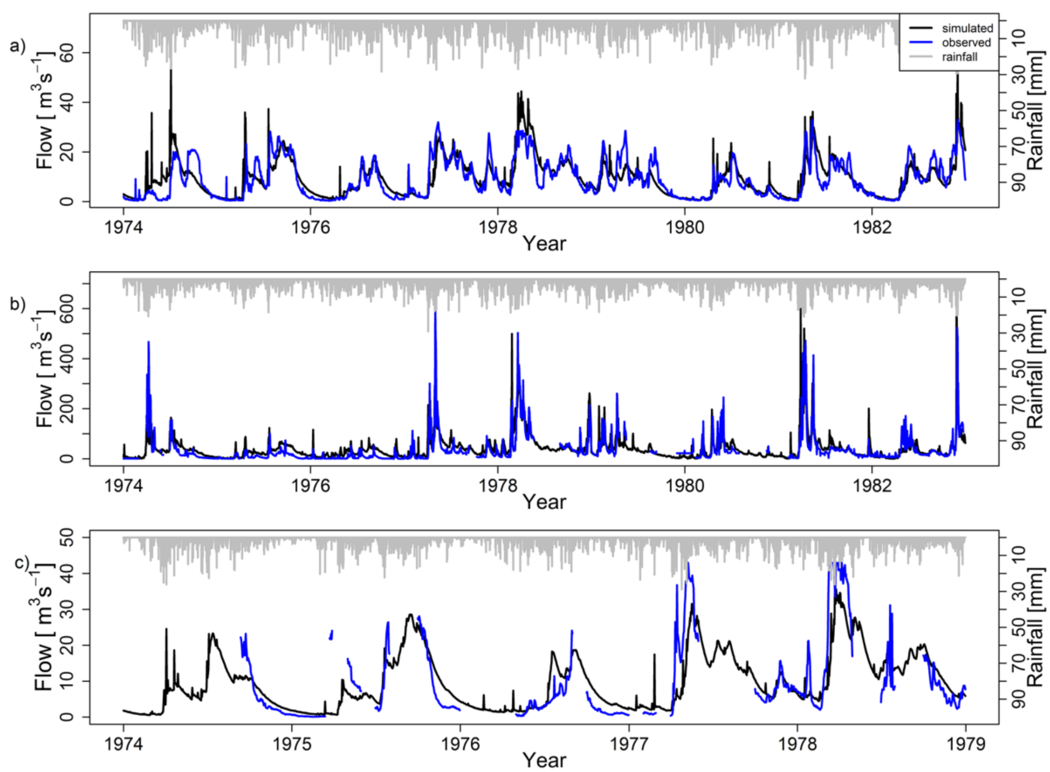

Figure 5.4: Daily hydrographs for observed and simulated streamflow of: a) Nyangores River at Bomet, b) Mara River at Mara mines and c) Amala River at Mulot

5.3.2 Impact of agroforestry on catchment water balance

Simulation results (Table 5.4) demonstrate that surface runoff, lateral flow, groundwater contribution to streamflow and the overall water yield decreased with increase in area under agroforestry. This was compensated by an increased rate of evapotranspiration. Surface runoff decreased by about 14%, 31% and 54% (Figure 5.5a) when the area of the watershed under tree cover was increased by 6.4%, 14.4% and 27.9% (Table 5.2) respectively. Similarly, groundwater contribution to streamflow decreased by about 5%, 11%, and 20% respectively for the three scenarios. The overall effect of the three scenarios on total water yield was a decrease by about 5%, 12%, and 22% respectively in that order of increasing tree cover. Meanwhile, evapotranspiration increased by 2%, 4% and 7% respectively. These results are consistent with the findings reported from paired catchment experimental studies (Brown et al., 2013; Zhao et al., 2012; Zhang et al., 2012; Scott and Lesch, 1997) and model simulation studies (Suarez et al., 2014; Githui et al., 2009) that have reported decrease in water yield and increase in evapotranspiration following establishment or increase of watershed tree cover.

Table 5.4: Water balance (in mm) of the Nyangores sub-watershed for the three agroforestry scenarios

	Base	S1	S2	S3
Precipitation		1429.6		
Surface runoff	29.7	25.5	20.6	13.8
Lateral flow	29.2	28.0	27.5	27.0
Groundwater flow (GwQ)	295.4	281.4	263.9	235.7
Revap	0.45	0.46	0.47	0.48
Total water yield	354.3	334.9	311.9	276.5
Evapotranspiration (ET)	1057.8	1076.4	1098.6	1133.1
Potential ET (PET)		1605.9		

Groundwater flow (GwQ) is the groundwater contribution to streamflow.

The decline in surface runoff can be attributed to increased infiltration (Brown et al., 2005; Benegas et al., 2014) and canopy interception (Ghazavi et al., 2008). Establishment of trees on land formerly under cultivated agriculture improves the infiltration conditions of the soil, thereby absorbing more rainfall and reducing the surface runoff. Field experimental study by Anderson et al. (2009), for example, reported significantly higher infiltration in the agroforestry buffer treatments compared with row crop treatments. Ketema and Yimer (2014) also reported higher infiltration for agroforestry treatments than for maize treatments for their study in Southern Ethiopia. Practising of intensive agricultural cultivation, as is the case currently in the upper Mara, continually degrades the soil and reduces its capacity to absorb rainwater mainly due to compaction of lower soil horizons, decrease in organic carbon and porosity (Recha et al., 2012; Bruijnzeel, 2004). Trees on the other hand, aid in the recovery of degraded lands (Udawatta et al., 2008; Lagerlöf et al., 2014). High organic matter, presence of live and dead roots, increased soil micro-fauna and enhanced macro-pore flow are some of the factors that improve soil infiltration after establishment of agroforestry (Ketema and Yimer, 2014; Udawatta and Anderson, 2008). However, it should be noted that soil infiltration capacity recovery may take some time (Bruijnzeel, 2004) and potential gains in water infiltration reported here may not be achieved immediately after the establishment of agroforestry (Brown et al., 2013).

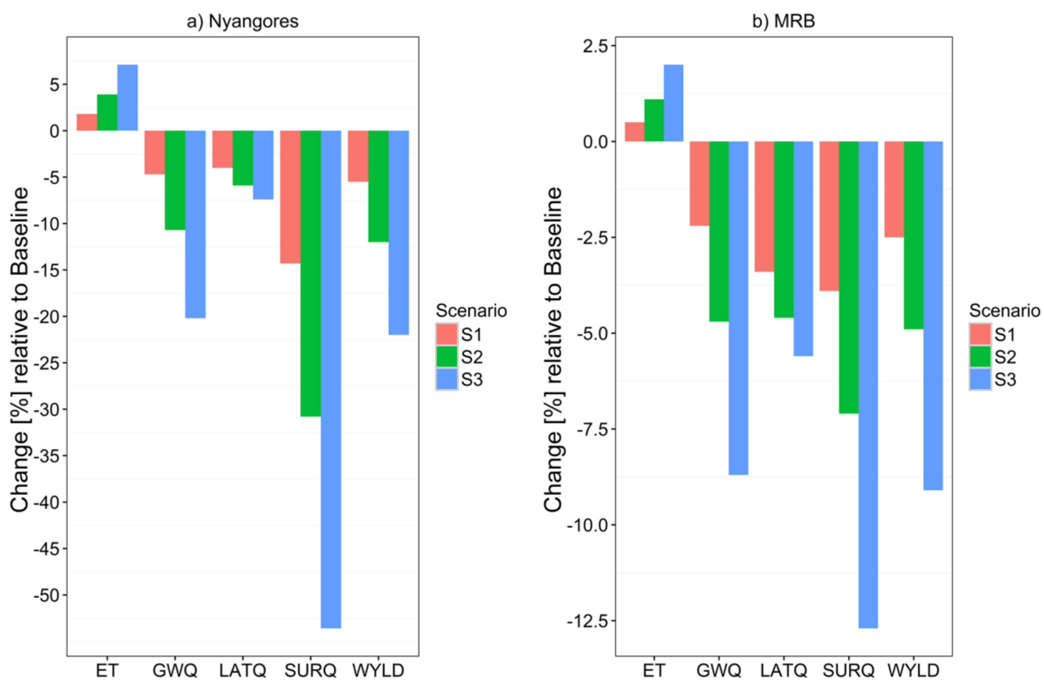

Figure 5.5: Relative impact of increasing area under agroforestry on water balance of: a) Nyangores sub-watershed and b) larger MRB. SurQ is the Surface runoff, LatQ is the lateral flow, GWQ is the groundwater contribution to streamflow, Total WYLD is the total water yield and ET is the evapotranspiration

Although there was increased infiltration for the agroforestry scenarios, which ideally increases recharge of aquifers, there was also a decrease in baseflow. This can be attributed to an increase in water extraction from the soil and aquifer by the trees. Trees often have deeper and more extensive rooting systems than other vegetation which enables them to extract groundwater to meet the evapotranspiration demand, especially during the dry seasons when the top soil is dry (Thomas et al., 2012; Doody and Benyon, 2011a; FAO, 2006; Calder, 2005; Benyon et al., 2006). A study by Pinto et al. (2014), for example, estimated that annual soil and groundwater contributions to tree transpiration were about 70% and 30%, respectively. However, during the dry summer months the groundwater contribution became dominant and rose to 73% of transpiration. Additionally, trees have higher aerodynamic roughness than crops that favour higher evapotranspiration rates (Calder, 2005). The differences in leaf, size, shape, thickness, anatomy and chlorophyll content between trees and other plants and even between trees species also affects the rate of transpiration (Muthuri et al., 2009). Consequently, increase in tree cover through agroforestry also increases water use in the watershed in form of evapotranspiration. A study by Muthuri et al. (2004) in central Kenya found that water use in agroforestry systems was higher than for treatments under only maize cultivation. The decrease in groundwater in shallow aquifers, due to increased uptake by trees, decreases the water available and the amount released to the streams as baseflow (Adelana et al., 2015; Fan et al., 2014; Doody and Benyon, 2011b). Generally, the change in baseflow may be either positive or negative depending on the water budget in the aquifer storage (Bruijnzeel, 2004). If the incoming water, as a result of improved infiltration, surpasses the extra water removal by trees, then the extra storage may lead to increase in baseflow. The reverse is also true in the case of negative change in aquifer storage as was the case in our study (Brown et al., 2005; Bruijnzeel, 2004). The overall water yield, which is essentially a summation of surface runoff, lateral flow and groundwater contribution to streamflow, also decreased with an increase in the area under agroforestry.

The water balance results (Table 5.4) are based on past climatic conditions (1980-1990). Because the base scenario was based on current land-use conditions (2014 land-use map), the changes in climate between the 1980's and 2014 may slightly affect the absolute values of the water balance. The changes are however, expected to be minimal. For the upper Mara, Mwangi et al., (2016b), estimated that climate variability only contributed about 2.5% increase in streamflow for Nyangores sub-watershed in the last half a century, the rest being contributed by land-use changes. No major changes, however, are expected on the relative results obtained for the simulation of agroforestry (Figure 5.5), because all the scenarios were assessed based on the base scenario (i.e. same climatic conditions between base scenario and all the other scenarios).

Similarly, climate change may as well affect the absolute values of water balance but not the relative changes (percentage change in water balance) due to implementation of agroforestry. Mwangi et al. (2016b) estimated that climate change would cause a 15% increase in streamflow (over the next 50 years) in the upper Mara watershed, which is indicative of how the absolute values of the water balance might change. The change in individual water balance components might, however, not be linear due change in climate seasonality (Dessu and Melesse, 2013).

5.3.3 Impact of spatial scale

For the larger MRB, surface runoff decreased by about 4%, 7% and 12.5% respectively for the three scenarios in the order of increasing area under agroforestry (Table 5.5; Figure 5.5b). The groundwater contribution to streamflow and the water yield similarly decreased by 2%, 4.5% and 8.5%, and 2.5%, 5% and 9% respectively for each of the three scenarios. The evapotranspiration however increased by approximately 0.5%, 1%, and 2% (Figure 5.5b). The results illustrate a similar trend as that of the Nyangores sub-watershed (Figure 5.5a) which can be attributed to similar causes. The only difference is in the magnitude of the relative changes. For all the water balance components, the relative change (impact of agroforestry) was larger at Nyangores sub-watershed compared to the larger MRB. This can be attributed to the differences in the ratio of area simulated with agroforestry to the total sizes of respective watersheds (Brown et al., 2005; Bruijnzeel, 2004). The proportion of watershed areas simulated with agroforestry were 1.8%, 3.3% and 6% of the watershed area respectively for the three scenarios for the MRB compared with 6.4%, 14.4% and 27.9% respectively for Nyangores sub-watershed. It is therefore apparent that watershed scale has a profound effect on the impact of agroforestry on watershed hydrology. Comparison of the relative impact of the ratio of watershed under agroforestry on water yield between the two watersheds, reveals an interesting effect of scale (Figure 5.6). It can be seen that although the impact of each of the agroforestry scenarios on water yield was higher for Nyangores sub-watershed, the slope was higher for the MRB than for Nyangores. This may have been caused by climate variability within the MRB (Brown et al., 2005). From Tables 5.4 and 5.5 it can be seen that whereas the average precipitation and potential evapotranspiration are the same across the three scenarios in each of the two watersheds, the values are different for the two. The average rainfall is higher for the upstream Nyangores sub-watershed (1430 mm) than for the larger MRB (1045 mm). This is because the lowlands (Maasai Mara-Serengeti region) experience lower rainfall compared with the upper Mara (Mau Forest). Average temperatures however, are higher in the lowlands than highlands and consequently the potential evapotranspiration is slightly higher for the larger Mara (1629 mm) than for Nyangores (1605 mm). This implies that generalisation or extrapolation of the impact of agroforestry (or of any other forest-related land-use change) of a small catchment to the larger watershed, may not be practical without considering the effect of climate variability within the watershed (Brown et al., 2005).

Table 5.5: Water balance (in mm) of the MRB for the three agroforestry scenarios

	Base	S1	S2	S3
Precipitation		1044.6		
Surface runoff	23.8	22.9	22.1	20.8
Lateral flow	10.3	9.9	9.8	9.7
Groundwater flow (GwQ)	106.1	103.8	101.2	96.9
Revap	124.4	123.9	123.5	122.8
Total water yield	140.1	136.6	133.2	127.3
Evapotranspiration (ET)	750.9	755.0	758.8	765.6
Potential ET (PET)		1628.9		

Groundwater flow (GwQ) is the groundwater contribution to streamflow.

5.3.4 Implication for water resources management

The main finding of this study is that agroforestry would increase water demand and hence evapotranspiration and reduce the water yield (streamflow) of the Mara River. Reduced flows may be a concern by water managers who are tasked with managing the resources against an increasing demand (Dessu et al., 2014). However, these findings should be viewed within the broader context of environmental services provided by agroforestry. This is necessary because in the last few decades there has been a paradigm shift on how water resources should be managed (Sivapalan et al., 2012). Integrated Water Resources Management (IWRM) has now been accepted worldwide as an effective management approach of water resources (UNEP, 2010; GWP, 2000). IWRM advocates for a holistic approach in water management where water, land and other resources (e.g. forestry) are managed in an integrated manner- because they are interlinked (Mwangi et al., 2015a). Agroforestry, for example, additionally provides other environmental services e.g. soil erosion control, provision of wood products such as timber and fuelwood, carbon sequestration, modification of microclimate (Ong et al., 2006; Nair, 1993). Soil erosion control is directly related to the findings reported here. The decrease in surface runoff due to agroforestry as reported in this study would consequently reduce soil erosion which is still a major problem in the MRB (Defersha and Melesse, 2012; Defersha et al., 2012; Kiragu, 2009). Reduced soil erosion would essentially reduce loss of top fertile soils in farmlands and hence control decline in land productivity for improved crop production. Decline in land productivity in the upper Mara has led to increased encroachment of the Mau forest by the local communities whose main economic activity is subsistence farming (Mati et al., 2008). Reduction in soil erosion would also minimize sedimentation in the rivers and thus improving the water quality. This is very important because the majority of people living in the watershed consume the water directly from the stream without any form of treatment (Ngugi et al., 2014; Dessu et al., 2014). For the few who live in towns within the watershed and who have the privilege of using treated water, reduced sediment loads would lower water treatment costs. Another key benefit of agroforestry is the provision of timber and fuelwood which would lower the pressure on the native forests. In Kenya, about 89% of people living in rural areas rely on fuelwood for their energy needs (World Resources Institute, 2007; Nyaga et al., 2015) which demonstrates the importance of agroforestry in the livelihoods of rural communities. Agroforestry would also be a means of restoring back some of the degraded parts of the watershed that was initially under forest.

It is also worth mentioning that the results reported here are based on annual averages. Water resources management should go beyond the annual averages and consider the intra-annual flows. This is because streamflow seasonality is a key determinant of water availability (Hoekstra et al., 2012) particularly for an unregulated river like Mara (Young, 2014). The River Mara is only 395 km long from the source to its mouth in Lake Victoria. This means it only takes a few days for water from the headwaters to drain in the Lake and therefore most of flood water especially in the two wet seasons ends up in the Lake and this may still be the case even in case of implementation of agroforestry. Flood water harvesting for the Mara would therefore be a very practical management strategy to ensure temporal distribution of water availability throughout the year.

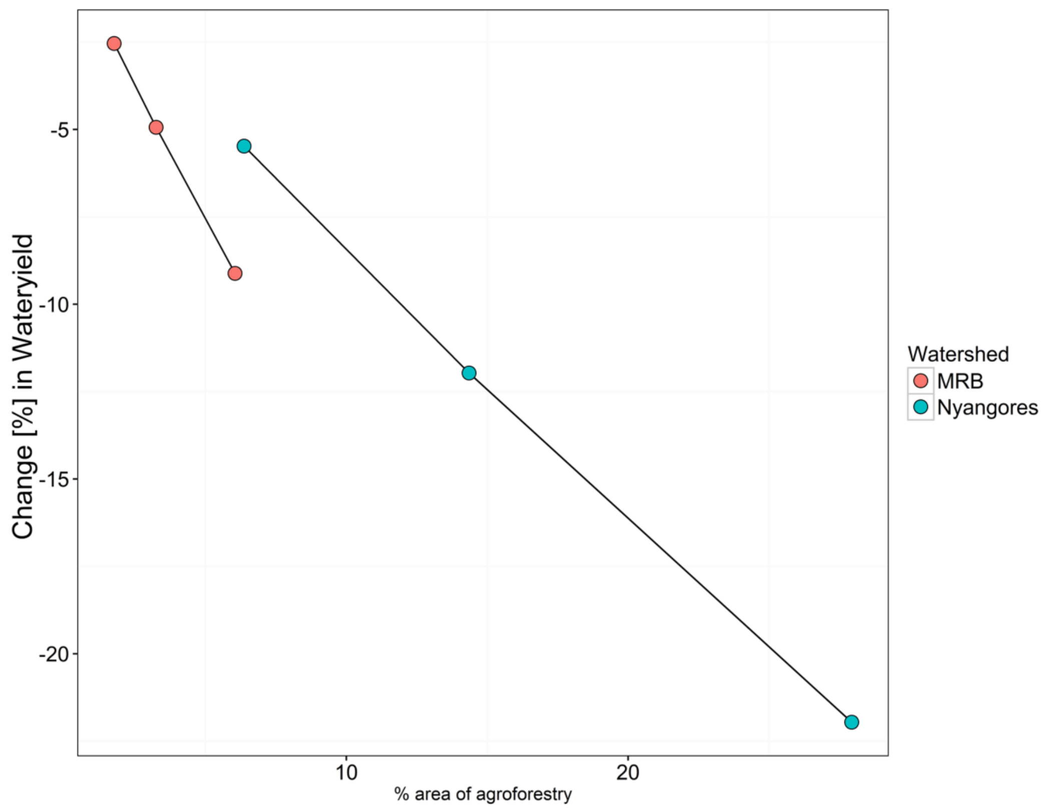

Figure 5.6: Relationship between ratio (%) of watershed simulated with agroforestry and change in water yield for MRB and Nyangores

Our study has shown that the reduction in mean streamflow, due to implementation of agroforestry on MRB, would be higher on the Kenyan side of the Mara compared to the Tanzanian side. Integration of management of trans-boundary basins is also emphasised in IWRM, therefore a more holistic view of watershed management in the MRB is required. Our findings viewed under the lens of IWRM, would therefore provide crucial information for watershed management of MRB. The three scenarios further provide some guidelines on trade-offs that can be made between streamflows and other environmental services especially by the Kenyan government which is keen to increase tree cover of the heavily deforested upper Mara basin and Mau Forest in general (GoK, 2009).

At a global level, the SWAT model is increasingly getting wide application in land-use and water resources studies (Gassman et al., 2010). Because agroforestry is also a common land-use practice worldwide, especially in tropical Africa, Asia and America (World Agroforestry Centre, 2009), there is a need to provide ways/methods of modelling agroforestry in SWAT. We have provided a simple approach using the current model structure, with good results. However, more is required to make the model structure flexible to enable modelling of different agroforestry systems e.g. allow intercropping in the same HRU.

5.4 Conclusions

The SWAT model was used to simulate the impact of agroforestry on the hydrology of the MRB. Prior to simulation of agroforestry scenarios, the model was successfully parameterized, calibrated and validated. We have provided a simple approach for simulating agroforestry in SWAT using the current model structure. We note however that more model structure flexibility is required to incorporate different agroforestry systems. We provide a simple way in which the model can reasonably simulate tree growth in the tropics without changing the source code. Though simple, this kind of parameterization, which involves adjusting the minimum LAI and fraction of tree heat unit to initiate growth, was considered better than the use of the default parameters that are better suited for temperate regions. Use of this approach of parameterization can greatly improve SWAT land-use modelling in tropical countries of the world.

Model simulation scenarios showed that agroforestry would generally reduce surface runoff, lateral flow, groundwater contribution to streamflow and the water yield while evapotranspiration would increase. The relative change in the water balance components was proportional with increase in area under agroforestry. The decrease in surface runoff was mainly attributed to improved water infiltration conditions offered by the trees. Overall water yield decline was attributed to extra water use by trees which extract water from shallow aquifer storage owing to their deep rooting system and also transpire more as a result of larger aerodynamic conductance.

Spatial scale was found to have a significant role in determining the magnitude of change in hydrology; the impact of agroforestry was bigger for the smaller up-stream Nyangores sub-watershed compared with that of the entire MRB. This shows that the impact on hydrology is directly related to the fraction of the watershed implemented with agroforestry. It was also found that the slope of change of water yield with increase in tree cover was different for the MRB compared to that of one of its upstream sub-watershed (Nyangores). This was attributed to the spatial variability of climate within the MRB. This implies that generalization or extrapolation of effect of agroforestry (or any other change in tree cover) from small to larger watersheds may not be accurate without eliminating or taking into account climate variability within or between watersheds. This information is particularly important for the scientific community working on small experimental study sites with an aim of extrapolating the results (or modelling) to large watersheds.

We conclude that these findings would be more beneficial to water resources managers when viewed from a broader perspective of IWRM. Agroforestry has many other related ecosystem services e.g. soil erosion control, which is directly related to our findings of reduction in surface runoff. Reduced surface runoff, and by extension soil erosion control, may also have multiplication of other benefits such as drinking water quality improvement and enhanced crop production for the subsistence farmers in the watershed. Owing to the high levels of competition for land between forestry and crop production in the basin, the results of the three agroforestry scenarios which are based on tree cover increment, may be used as a guideline to assist water resource managers and policy makers in making practical trade-offs between change in water yield and other benefits of agroforestry.

6 Chapter six: Synthesis and conclusions

6.1 Land use change

Land use change was assessed by the intensity analysis approach (Aldwaik and Pontius, 2012) using data from land use maps generated from Landsat satellite images taken between 1976 and 2014 in four consecutive intervals. Forest loss and expansion of agriculture were found to be on the rise. Forest reduced from about 20% of the study area in 1976 to about 7.5% in 2014. Agriculture (small scale and large scale) increased from about 6.5% of the watershed in 1976 to about 21% in 2014. Swap change accounted for more than 50% of overall change during the entire study period. Swap change is a situation where a land use category loses at one point of the study area but gain of equal size occurs in another part of the study area. This implies that the observed net changes underestimated the total land use change, which was found to be more than double the net changes. The net change in closed forest between 1995 and 2003, for example, was only 10% of the total change (of closed forest). Swap changes accounted for 90% because forest loss was accompanied by forest gain (regrowth) in other parts of the watershed. Results from intensity analysis of land use change showed there was systematic intensive conversion of forest to small scale agriculture and conversion of rangeland to largescale mechanized agriculture throughout the study period.

The high rate of deforestation in the watershed was attributed to encroachment and a series of excisions particularly on the Mau forest which has occurred since Kenya's independence in 1963 (GoK, 2009). The local communities have been progressively encroaching into the Mau forest reserve over the years to the extent that the government has started evicting them from the forest reserve. This process has however been faced with some challenges including political interference, making it slow. The encroached parts of the Mau forest are used for small scale agriculture, and hence the observed simultaneous rise in agriculture. The increase in largescale mechanized agriculture was attributed to change in land tenure of the rangeland (from communal to private) (Kimani and Pickard, 1998). Privatization of the rangelands enabled owners to directly lease out their land for large-scale farming (especially of wheat) to supplement their income from pastoralism (Thompson and Homewood, 2002).

6.2 Change in streamflow due to land use change and climate variability

The pertinent question dealt with in chapter four was whether there was a change in watershed hydrology. Streamflow is the main component of catchment water balance regularly measured in many watersheds and its analysis can give more insights into the dynamics of watershed hydrology. Due to data availability and quality challenges, only data from one of the upstream gauging stations of the Mara River (Nyangores at Bomet) was used for the analysis. The results from this headwater sub-watershed were however considered to be indicative of change in hydrology of the entire MRB. Results show a significant increasing trend of annual streamflow (1965-2007) with a slope of about 4.75 mm/year. Land use change contributed about 97.5% of the change while the rest of the change (2.5%) was caused by climate variability. The high contribution of change in streamflow by land use change was mainly attributed to the high level deforestation observed in the watershed (chapter three). Results from land use change analysis (chapter three) indicate that there has been consistence conversion of forest into agriculture. Trees are generally known to use more water (transpire) than most vegetation. Trees also have deeper and more extensive roots which are able to extract groundwater at relatively larger depths (Bruijnzeel, 2004). Change from forest to agricultural crops would therefore reduce vegetation water use and thus the observed deforestation may be the main reason for increased streamflow. Deforestation and conversion to agriculture may as well reduce rainwater infiltration due to soil degradation. This may essentially increase surface runoff and reduce groundwater recharge. For this study however, separation of streamflow into baseflow and quick runoff showed that baseflow had a similar increasing trend as streamflow. This supports the reasoning that reduced extraction of soil water and groundwater by vegetation is mainly responsible for increase the in streamflow. The ‘extra’ groundwater after deforestation is released to streams as baseflow.

The minimal impact of climate variability (2.5% of change in streamflow) was attributed to counter effect of increased rainfall and increase in mean temperature. Increase in temperature raised the atmospheric water demand of the watershed (potential evapotranspiration). Thus, the expected impact of increased rainfall on streamflow (increase of 24.2%) was reduced by 21.6% due to the extra atmospheric demand. This information was used to develop a simple runoff sensitivity equation to predict impact of climate change on streamflow. It was predicted that climate change would have a net increase in mean annual streamflow of 15% in the next 50 years.

6.3 Impact of agroforestry on watershed hydrology

Agroforestry is one of the feasible and practical ways of increasing forest cover in some of the parts of the watershed previously under forest but currently under intensive cultivation. The SWAT model was used to investigate the impact of agroforestry on the watershed hydrology. Model calibration results showed that the model was able to simulate the rainfall-runoff processes of the watershed. The model performance was considered satisfactory based on Nash-Sutcliffe efficiency (NSE) and Kling-Gupta efficiency (e.g. NSE values of 0.78 and 0.79 were obtained respectively for calibration and validation of streamflow of Mara River at Mara mines gauging station). It was found that implementation of agroforestry in the watershed would reduce surface runoff, baseflow and the overall water yield (streamflow); evapotranspiration would however increase. Reduction in surface runoff was attributed to increased infiltration and canopy interception expected after establishment of agroforestry (Brown et al., 2005; Ghazavi et al., 2008). The decline in baseflow was attributed to increase in water extraction from soil and aquifers by trees. This is opposite of the observed impact of deforestation in the watershed (chapter 4). The results imply that any extra recharge, due to enhanced infiltration brought about by agroforestry, would be outweighed by extra groundwater extraction by the agroforestry trees i.e. increased evapotranspiration. The observed changes (in surface runoff, baseflow, water yield and evapotranspiration) were proportional to increase in size of the watershed simulated with agroforestry. For example, an increase of tree cover of about 2% of the watershed through agroforestry would decrease the water yield (streamflow) by about 2.5% while a decrease of about 9% of streamflow would be expected if the tree cover is increased by 6% of the watershed through agroforestry. Since decrease in streamflow may be a concern to watershed managers, the results of the three scenarios of agroforestry simulated with increasing levels of tree cover may be useful for decision making on the level of tradeoffs (between reduced streamflow and increase in tree cover) appropiate for the watershed.

The impact of agroforestry on streamflow was larger for Nyangores sub-watershed compared to that of the larger MRB. This was attributed to effect of spatial scale because relatively larger fraction of the watershed was simulated with agroforestry for Nyangores compared to that of MRB for each scenario (Chapter 5). It was however observed that the slope (rate) of change in water yield with increase in tree cover was higher for MRB than for Nyangores which was attributed to climate variability (rainfall and temperature) within the MRB. The mean rainfall for the upstream Nyangores sub-watershed is higher than the average rainfall for the MRB. The mean temperatures (and consequently potential evapotranspiration) are however higher for the lowers sections (and hence the average for MRB) than for Nyangores. This implies that the entire MRB has comparatively less available water (rainfall) and higher atmospheric demand (potential evapotranspiration) while its opposite for Nyangores; and thus the higher impact of water 'removal' by agroforestry for the MRB than for Nyangores. These findings are not only useful for planning of agroforestry in the Mara River Basin but also for the wider scientific community working on small watersheds and wish to extrapolate their results for larger basins. The findings imply that generalization or extrapolation of impacts of land use change (e.g. agroforestry and afforestation) on streamflow from small (experimental) study sites to larger watersheds need to take climate variability into account.

6.4 Conclusions and recommendations for watershed management

The Mara River Basin has undergone substantial change in land use over the last 40 years. Transitions from forest to small scale agriculture and from rangeland to largescale mechanized agriculture are dominant land use changes, which indicate intensification of deforestation and expansion of agriculture in the watershed. Swap change accounts for more than half of the overall change land use change which implies that overall land use change is more than double of net changes that have been previously reported.

Streamflow of the Mara River (as indicated by data from Nyangores tributary) has increased in the last half a century. The observed land use (particularly deforestation and intensification of agriculture) contributed about 97.5% of the change (increase) in streamflow. Climate variability (change in rainfall and temperature) contributed the rest (2.5%) of the change in streamflow. The minimal contribution of climate variability to change in streamflow was caused by counter effects of change in rainfall (increase) and temperature (increase). Increase in temperature increased atmospheric water demand (potential evapotranspiration) that reduced the gains in streamflow that would have been caused by increase in rainfall.

The SWAT model was capable of simulating the rainfall-runoff processes of the Mara River Basin (based on the model performance which was assessed using Nash-Sutcliffe efficiency and Kling-Gupta efficiency). SWAT simulation results suggested that implementation of agroforestry in the watershed would cause a reduction in surface runoff, baseflow and total water yield (streamflow) and an increase in evapotranspiration. Reduction in surface runoff was attributed to expected improvement of the soil infiltration properties of the currently degraded lands under intensive cultivation and increase in canopy interception by agroforestry trees. Decline in baseflow was attributed ‘extra’ water extraction from soil and groundwater through transpiration. The changes (in surface runoff, baseflow, water yield and evapotranspiration) were proportional to size of the watershed simulated with agroforestry. Three scenarios with increasing levels of tree cover through agroforestry were simulated. The findings from these scenarios may be used for selection of the practical size of land that can be simulated with agroforestry, considering the change in water balance of the watershed. Climate variability within the basin has a profound effect on the impact (change) of agroforestry on catchment water balance. The difference in average rainfall and temperature between the entire MRB and one of its upstream sub-watershed (Nyangores) caused the rate of change of streamflow with increase in size of watershed simulated with agroforestry to be higher for MRB than for Nyangores.

Specific recommendations are given in respective chapters of this dissertation. The focus of this section is general recommendations of the entire study, particularly on watershed management and conservation. Regarding deforestation, it is recommended that the Government of Kenya should put more effort in arresting further deforestation of the Mau forest. This study attributed the observed deforestation to progressive encroachment and excision of the forest reserves. Forest excisions were initiated by the government in the past political regimes and thus it would be easy to stop further excisions provided there is political goodwill backed up by good policies and functional institutions. The results indicate that closed forests are first opened up before conversion to agriculture i.e. open forest is a transitional land cover between closed forest and agriculture. There is a possibility that timber and charcoal traders could be behind the opening up of forest, which then gives the local community easier access for cultivation and settlement. Monitoring and control of timber and charcoal business around the forest reserves may therefore be an effective strategy for limiting further encroachment into the forest reserve.

Land use change was found to be the main driver of change in hydrology of the Mara River and therefore watershed managers should prioritize and place more emphasis on reversing the degradation of the watershed. Agroforestry is recommended as a practical management and conservation strategy for the watershed that would also raise the tree cover, as desired by the

Government of Kenya. Agroforestry would also provide extra income to farmers through sale of timber and charcoal which would in turn reduce illegal logging of the remaining forests. Furthermore, agroforestry would also provide fuelwood to famers, most of whom depend on fuelwood for their energy requirements. These additional benefits would make it easier for adoption and acceptance of agroforestry as a conservation measure. The results (impact on water balance) of the simulated agroforesty scenarios (based on tree cover increment) may be used as a guide to determine the additional size of the watershed that may practically and sustainably be to put under tree cover. Tree species with low transpiration (water uptake) should be considered for agroforestry because the findings of this study show evapotranspiration is the major process that would affect agroforestry impact on catchment water balance. It is also recommended that more agroforestry should be planned for high elevations at the headwaters of the basin where rainfall is relatively higher and atmospheric water demand (potential evapotranspiration) is lower compared to lowlands, thereby leading to relatively lower change (reduction) change in water yield.

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