A SPATIALLY EXPLICIT APPROACH TO DETERMINE HYDROLOGY, EROSION AND NUTRIENTS DYNAMICS IN AN UPSTREAM CATCHMENT OF LAKE VICTORIA BASIN

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DISSERTATION

to obtain
the degree of doctor at the University of Twente,
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prof.dr. H. Brinksma,
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by

Ejiet John Wasige

born on 21 January 1972

in Tororo, Uganda

This thesis is approved by
Prof.dr. E.M.A. Smaling, promotor
Prof.dr. V.G. Jetten, promotor
Dr. T.A. Groen, assistant promotor

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Chapter 1

General introduction

1.1 Research background

In the past four decades, the global community has been concerned over the state of the world's land resources to sustain the ever increasing global population and worsening world food situation (FAO, 2011). These concerns culminated into the Brundtland Commission Report of 1987 (WCED, 1987), the FAO (1971) land degradation report, the 1992 held United Nations Conference on Environment and Development (UNCED) and the 2002 World Summit on sustainable development. The reports and conferences have resulted into a suite of international multilateral environmental agreements and organizations such as; The UN Framework Convention on Climate Change (UNFCCC), the UN Convention to Combat Desertification (UNCCD), Convention on Biological Diversity (CBD), The 'Clean Development Mechanism' (CDM), Reducing Emissions from Deforestation and Forest Degradation "plus" conservation, the sustainable management of forests and enhancement of forest carbon stocks (REDD+), Intergovernmental Panel on Climate Change (IPCC), Millennium Development Goals (MDGs). The Agenda 21, the legacy of the 1992 UN Conference of Environment and Development in Rio, describes a series of environmental issues to be addressed and avenues to be followed to move closer to "sustainable development". Soil management issues appear prominently in Agenda 21's priorities for sustainable land management, protecting the atmosphere, sustainable mountain development and combatting deforestation and desertification (Sanchez, 1994; Smaling et al., 1996). Chapter 14 specifically deals with sustainable agriculture and rural development. Program area J deals with "sustainable plant nutrition to increase food production", and singles out Sub-Saharan Africa (SSA) as the subcontinent that is losing soil fertility at an alarming rate (Smaling et al., 1996). In 2012, the UNCED was convened as a 20 year follow-up to the historic 1992 UN Rio Conference. The 2012 HDR (human development report) shows that inequality and deteriorating environmental conditions will together pose obstacles to progress in Africa and across the globe (HDR, 2011).

The major global environmental problems include climate change and increased greenhouse effect caused by increasing levels of greenhouses gases, the extinction of rare animals and plant germplasm, the negative impact of deforestation and land degradation, and decreasing food security. Land degradation remains an important global issue for the 21st century because of its diverse effects on agronomic productivity, emissions of greenhouse gases (GHGs) into the atmosphere, environmental quality, food security and the quality of human life (Lal, 1998, 2009; Eswaran et al., 2001; FAO. 2011; Smaling et al., 2012). Land degradation is caused by natural and anthropogenic causes. Natural causes include the inherently low capacity of some ecosystems to provide goods and services after minimal disturbance. These include climatic disturbances such as droughts, inherent climatic factors determining the capacity to generate biomass and provide ground cover and biodiversity, soil and terrain related causes such as slope and soil vulnerability to water and wind erosion, and water availability (Nachtergaele et al., 2012). Human-induced land degradation is the major destructive factor for natural resources in the world, and is recognized as a key issue for conservation in the 21st century (Reich et al., 2000). Human-induced causes are largely determined by land use and land use changes (Nachtergaele et al., 2012). A number of direct causes are seemly natural but may have wholly or partly indirect human causes e.g. forest fires, floods, landslides and droughts as a result of human-induced land degradation and climate change. Behind the direct obvious causes of human-induced land degradation, there often exist other, more deeply rooted drivers that have to do with population pressure, poverty, lack of markets and infrastructure, poor governance, weak institutional frameworks and inadequate education. The principal direct causes of land degradation are deforestation and land clearing, leading to degraded conditions such as reduced vegetation cover, soil compaction and increased run-off, exposure to water and wind erosion, and consequent sedimentation and siltation in water bodies. These direct causes often result from inappropriate land use and land management (Lal et al., 1989; Nachtergaele et al., 2012).

1.2 The global assessments of land degradation

The global assessments of land degradation started more than 35 years ago (e.g., The Global Assessment of Human Induced Soil degradation (GLASOD) project (1987-1990)). Global estimates of the extent and severity of land degradation and vulnerability to degradation processes are alarming (Oldeman, 1994; Kaiser, 2004; Reich and Eswaran, 2004). There are few systematic measurements of its extent and severity land degradation (Burch et al., 1987; Lal, 1998; UNEP, 2007; Nachtergaele et al., 2012). There is insufficient awareness and urgency because degradation is, by political standards, a slow process and its effects are postponed because compensation is possible in various ways (e.g. expansion to new land or extra fertilizer use) and does not often generate political capital. National food security is not threatened by degradation of land quality, as food can be bought from other countries, but food self-sufficiency is vulnerable (Stoorvogel et al., 1993; Penning de Vries, 2000).

A lot of empirical data is available on physical soil degradation worldwide, but most of this data is plot- or site-specific and quite difficult to aggregate and interpret on a national, regional or global basis (Bindraban et al. 2012; Nachtergaele et al., 2012). Institutional, social economic and biophysical causes of land degradation have been identified locally in many case studies but these have not been inventoried systematically at watershed, district, national, or regional levels (Nachtergaele et al., 2012). Identifying long-term options for sustainable management of soil and water resources necessitates credible data on the state-of-the-soil and water resources and their impacts on soil productivity and environmental quality.

There have been attempts to remedy the situation of incoherent data available on land degradation. The Land Degradation Assessment in Drylands (LADA) project started with the general purpose of creating the basis for informed policy advice on land degradation at global, national and local level. This goal was to be realized through the assessment of land degradation at different spatial and temporal scales and the creation of a baseline at global level for future monitoring. Common methods for assessing land degradation as identified in the LADA approach (Koohafkan et al., 2003) include: expert judgment (e.g., GLASOD, (Oldeman et al., 1990)), remote sensing (e.g., Global Land Degradation Assessment (GLADA), productivity changes using crop performance indicators (e.g., Bai and Dent, 2006; Madrigal et al., 2003; Lobell et al., 2007), field monitoring (e.g., Wessels et al., 2007; Naseri, 1998; Rostagno et al., 1999), pilot studies at farm level (e.g., Okoba and Sterk, 2006; Chartier et al., 2009), soil nutrient budgets (e.g., Cobo et al. 2010; Smaling et al., 2012) and modeling with established models for soil erosion by wind and water (e.g., PESERA, WATEM-SEDEM, SPADS, WEPP, FuDSEM, LISEM, etc). GLASOD assessed not the full range of ecosystem goods and services but was limited to soil degradation evaluation at an average scale of 1: 15 million (Oldeman et al., 1990). The assessment was based on expert judgment by a formal survey, mostly by a single local soil expert in a country or region and suffered from inconsistence and lack of reproducibility (Bindraban et al. 2012). Nevertheless, the GLASOD inventory remains the first global assessment that made policy makers aware of the widespread extent and impact of land degradation (Nachtergaele et al., 2012). The LADA project was launched by the Global Environmental Facility (GEF) during August 2006 in Cape Town, South Africa, implemented by UN Environment Programme (UNEP) and executed by FAO. The goal of the LADA project was to generate up-to-date ecological, social, economic and technical information, including a combination of traditional knowledge and modern science, to guide integrated and cross-sectoral planning and management in drylands. Under the LADA project, two different global assessments were undertaken; the Global Land Degradation Assessment (GLADA) and Global Land Degradation Information System (GLADIS). GLADA identified the Normalized Difference Vegetation Index (NDVI) based vegetation greenness over the period 1981-2006. It defined critical areas as those where both the greenness and rain-use efficiency were declining. In contrast to the GLASOD, the method used was analysis of measured data and it did restrict itself to a well-defined time period. The results indicated that the decline in greenness affected areas where 1 billion people were living and would result in a net loss of about 35 million tons of carbon yr^{-1}. Most affected areas were in the tropical Africa south of the equator and south east Africa, southeast Asia, south China, north-central Australia, drylands and steep lands of Central America and the Caribbean, Southeast Brazil and the pampas, and the Boreal forest (Bai et al., 2008; UNEP, 2007). Results were criticized because, although the observations were objective, the statistical methods used were debatable and the method focused only on a single ecosystem parameter (biomass/carbon). GLADIS was based on assessment of the status and trends of ecosystem goods and services (biomass, soil health, water quantity and quality, biodiversity, economics, social and cultural), including the impact of these changes on the population living in these areas. The major changes to ecosystems globally were studied over a period of 15-25 years (Nachtergaele et al., 2012). At this scale local changes are masked and therefore this study is of limited use for local natural resource planning and management.

In reaction to the GLASOD that was limited to soils, the World Overview of Conservation Approaches and Technologies (WOCAT) which is an initiative of the World Association of Soil and Water Conservation (WASWC) was started in 1992. WOCAT is driven by the notion that there has always been a heavy focus on documenting degradation but too little on sustainable land management (SLM) practices and that there is a wealth of knowledge on technologies for prevention and mitigation of land degradation, and rehabilitation of degraded land. Likewise, traditional land use systems and local land management innovations have been inadequately documented or assessed for their combined benefits in terms of productivity, conservation effectiveness and sustainability (Schwilch et al., 2011). The mandate of WOCAT is to improve the knowledge base underlying sustainable land management (SLM), through gathering information on the application of SLM worldwide. The focus on SLM complements the technical approach to land management with social and economic dimensions (Hurni, 2000). The WOCAT approach is now a standardized tool for comprehensive documentation and evaluation of Soil and Water Conservation (SWC) practices. Other SLM efforts include the DESIRE (Decision Support System on strategies for sustainable land management at local and regional scales) project (2007-2012; http://www.desire-project.eu) that is developing and testing alternative strategies for desertification-vulnerable areas. Like WOCAT, DESIRE advocates an SLM approach based on inventories of local knowledge. The DESIRE project covers a wide range of problems from soil erosion by wind or water, to salinisation and droughts or flash floods. DESIRE project scientists are currently working in 17 study sites in 13 countries (from southern Europe, southern America, Africa, Russia and China) with an integrative participatory approach, in close collaboration with local stakeholders as well as having a sound scientific basis for the effectiveness at various scales. The Global Environment Facility (GEF) (2009) has been the largest development initiative fostering SLM as a strategic intervention through its land degradation focal area. SLM is considered in a comprehensive manner, aiming at a global systems approach with mutual benefits for local people and the global environment (Stocking, 2009). GEF is currently developing tools to monitor and assess SLM progress in its project portfolio through its knowledge from the land initiative (Schwilch et al., 2011).

The GLASOD, LADA and WOCAT approaches to land degradation assessments have been criticized for being qualitatively based on land use systems, too coarse (global or regional scale) and expert judgment made from one spot/ case study. What is missing in these assessments are small scale, physiographic or toposequence approach that may reveal the quality of what is under the degrading land use systems; the potential loss of carbon, biodiversity loss, and nutrient depletion. Many of the SLM technologies have been applied and tested in the field or on experimental sites to assess their biophysical effectiveness, but assessments of their cost-effectiveness, impacts on ecosystem functions and services, on overall ecosystem integrity and on the economy are still weak (Bainbridge, 2007; Carpenter et al., 2006). One of the main tasks for scientific support of SLM is to produce evidence of its impact on natural resources and to assess the implications from such impacts on society, the economy and policy (Hurni et al., 2006). This is urgently needed, as it is now widely acknowledged that SLM has potential major global benefits, not just to counter land degradation but to simultaneously sustain ecological functions, contribute to biodiversity conservation and as a tool in the mitigation of, and adaptation to, climate change (e.g. Gisladottir and Stocking, 2005; Cowie et al., 2011; Schwilch et al., 2011).

1.2.1 Land degradation by water erosion

Land degradation due to accelerated erosion is a serious global issue because soil resources of the world are finite, nonrenewable at the human-time scale and sensitive to land misuse and soil mismanagement, especially in ecologically sensitive ecoregions such as the tropics (Lal, 1998). On the basis of area coverage by water erosion, Oldeman (1994) reported that the global extent of water erosion at the continental scale is in the order of Asia > Africa > South America > Europe > Oceania > North America > Central America. It is estimated that one sixth of the surface land is affected by accelerated water erosion (Schroter et al., 2005). More than 56% of land degradation is a result of water erosion, followed by wind erosion (28%) and together with sedimentation by these agents, this kind of land degradation is causing long-term reduction in crop yields (FAO, 1994). Soil erosion takes away the top soil, removing fertile land from agricultural use. Controlling erosion is therefore essential for minimizing loss of productivity, sedimentation and water quality degradation. Yield reduction in Africa due to past soil erosion may range from 2 to 40%, with a mean loss of 8.2% for whole the continent (Eswaran et al., 2001). Globally, an estimated annual loss of 75 billion tons of soil costs the world about US$ 400 billion, or approximately US$ 70 per person per year (Eswaran et al., 2001). The hot spots of erosion-induced soil degradation in Sub Saharan Africa (SSA) (Table 1.1) are mostly in the areas of the Sahel, the Highlands of Eastern and Central Africa and Ethiopia, and the Lake Victoria basin (World Bank, 1996).

Table 1.1: Global Hot Spots of Erosion-Induced Soil Degradation (modified from Lal, 1998)

Region	Cause
Asia	
South and West Asia: Lower and middle ranges of Himalayas	Land clearing and farming of marginal lands Conversion of rangeland in west Asia to cereal production
East and Southeast Asia: Steeplands in southern China and SE Asia	Intensive cropping Lack of conservation farming
Africa	
West/Central Africa: South eastern Nigeria	Conversion of shifting cultivation to intensive cropping Poor drainage outlet from roads and buildings
Sahel	Foot paths and cattle trails to rivers Low input and subsistence agriculture Resource poor farmers
Highlands of Eastern and Central Africa: Ethiopia, Kenya, Uganda, Rwanda, Burundi	Intensive cropping without conservation effective measures High stocking rate
North west Africa	Mechanization with inappropriate plowing techniques Lack of conservation effective measures
Latin America	
Central American highlands	High stocking rate, steepland cultivation without conservation measures
Andean hills	Inappropriate land use and soil mismanagement
Haiti and Dominican Republic	Cultivation of steep lands without conservation measures
Cerrados	Intense mechanization for row crop farming
Pacific	
Australia: Semiarid and subhumid regions	Intensive grazing, Grain crop cultivation
Oceania	Steepland cultivation, Subsistence agriculture

Accelerated soil erosion in Africa has been attributed to numerous causes (Lal, 1989), including tropical deforestation, land clearing and inappropriate farming practices in the highlands of east and central Africa (Lal, 1981; Worch et al., 1989), land clearing and inappropriate land use and soil mismanagement in the lake Victoria basin (Isabirye, 2005), agricultural intensification and lack of maintenance of traditional hillside terraces in Rwanda, Burundi and Uganda (Conelly, 1994; Roose and Ndayizigiye, 1997;

Tukahirwa,1996; Bagoora, 1998), low inherent soil fertility, and low available water holding capacity (AWC) (Zaongo et al., 1994).

Soil erosion rates vary globally among ecosystems (Table 1.2). It is difficult to validate the credibility of global erosion rates reported for predominant ecoregions of the world often due to lack of information on the methodology of data collection. For most of the world the erosion data is inadequate (Lal, 1990; Boardman, 2006). The available estimates are tentative and subjective and need to be improved by remote sensing, GIS and other modern techniques (Lal, 2001; Jetten et al., 2003, Boix-Fayos et al., 2006). Worldwide, erosion on cropland averages about 30 t ha^{-1} yr^{-1} and ranges from 0.5 to 400 t ha^{-1} yr^{-1} (Pimentel et al., 1995). El-Ashry and Ram (1991) observed that in some parts of semiarid Africa, as much as 450 t ha^{-1} of soil erodes annually. Severe soil erosion rates (greater than 50 ton ha^{-1} yr^{-1}) in Africa have been reported in the croplands of Rwanda, Burundi, Ivory Coast, Madagascar, Tanzania, Uganda, Niger and Senegal (Table 1.2). The risk of soil erosion and attendant land degradation are more severe in hot and dry than in cold and moist/ humid climates (Stewart et al., 1990). Soil erosion rates are increasing in the tropics and subtropics especially on marginal agricultural lands managed with low input and resource-based production systems (Lal, 1998). There is a combined effect of high rainfall intensities, soil properties, and poor agricultural practices. Rates of erosion are particularly high in steep areas (slope > 10°).

1.2.2 Land degradation by soil nutrient depletion

A part from soil erosion, soil nutrient depletion is another widespread soil degradation phenomenon that occurs largely as a consequence of soil erosion and nutrient mining. It is generally top soil, in which most soil nutrients are present, that erodes fastest. But also poor management practices such as slash and burn, nutrient export by removal of harvest and crop residues without replenishing nutrients is a cause of nutrient depletion (Nachtergaele et al., 2012; Smaling et al., 2012). The process of soil nutrient depletion is a potentially serious threat to world food security (Lal, 2009; Smaling et al., 2012). This is evident in the long-term decline in crop yields under conditions of low-input and unbalanced fertilization in many parts of Africa (FAO/UNDP/UNEP/World Bank, 1997; Smaling et al., 2012). It's an important concern directly linked to food insecurity and poor human health due to inadequate nutrition in developing countries (Lal, 2009). Other, more indirect, consequences of nutrient depletion include biodiversity losses, sedimentation within watersheds, and pollution of water bodies (Sanginga and Woomer, 2009). Nutrient depletion is now considered the chief biophysical factor limiting small-scale farm production in Africa (Smaling et al. 1993; Sanchez et al., 1997; Drechsel et al., 2004).

Table 1.2: Global soil erosion rates in different ecosystems

Country Site	erosion rate (t/ha/year)	Comments	
United States	18.1*	average, all cropland	Pimentel et al., 1986
midwest deep loess hills (IA and MO) and southern high plains (KS, NM, OK, and TX)	35.6-51.5*	Major Land Resource Areas	Pimentel et al., 1986
China	11-251	average, all cultivated land	Pimentel et al., 1986/ Lal et al., 1989
Yellow River Basin	100	middle reaches, cultivated rolling loess	Pimentel et al., 1986
India	28-75	cultivated land	
Deccan black soil region	40-100		Pimentel et al., 1986
Java, Indonesia	43.4	Brantas River Basin	Pimentel et al., 1986
Belgium	10-25	Central Belgium, agricultural loess soils	Lal et al., 1989
East Germany	13	1000-year average, cultivated loess soils in one region	Pimentel et al., 1986
El Salvador	19-190	Acelhuate Basin, land under basic grains production	Pimentel et al., 1986
Jamaica	90	cropland	Lal et al., 1989
Guatemala	200-3600	corn production in mountain region	Pimentel et al., 1986
Guatemala	5 -35	cropland	Lal et al., 1989
Ecuador	210-564	cropland	Lal et al., 1989
Thailand	21	Chao River Basin	Pimentel et al., 1986
Nepal	40	cropland	Lal et al., 1989
Burma	139	Irrawaddy River Basin	Pimentel et al., 1986
Venezuela and Colombia	18	Orinoco River Basin	Pimentel et al., 1986/ Lal et al., 1989
Argentina, Paraguay and Brazil	18.8	cropland	Lal et al., 1989
Peru	15	cropland	Lal et al., 1989
Ethiopia	20-34	Cropland/ Simien Mountains, Gondor region	Pimentel et al., 1986/ Lal et al., 1989
Madagascar	25-250	nationwide average	Pimentel et al., 1986/ Lal et al., 1989
Nigeria	14.4	Imo region, includes uncultivated land	Pimentel et al., 1986/ Lal et al., 1989
Benin	17-28	cropland	Lal et al., 1989
Burkina Faso	10-20	cropland	Lal et al., 1989
Lithoso	40	cropland	Lal et al., 1989
Guinea	17.9-24.5	cropland	Lal et al., 1989
Kenya	5-47.1	cropland	Lal et al., 1989
Zimbabwe	50	cropland	Tagwira,1992
Senegal	14.9-55	cropland	Lal et al., 1989
Niger	35-70	cropland	La et al., 1989
Uganda	17.0 – 129	cropland	Tukahirwa,1996; Bagoora, 1998
Tanzania	10.1-92.8	cropland	Lal et al., 1989
Tanzania	72 -120	Usambara Mountains	Lundgren, 1980
Papua New Guinea	6-320	cropland	Lal et al., 1989
Ivory Coast	60-570	cropland	Lal et al., 1989
Rwanda/ Burundi	300-700	Bare soil	Roose & Ndayizigiye, 1997
Rwanda/ Burundi	20-150	cropland	Roose and Ndayizigiye, 1997

*Indicates combined wind and water erosion, all others are water only

Nitrogen (N), phosphorus (P) and Potassium (K) nutrient balances are considered the main nutrients to investigate because they are included in soil quality studies. These elements are indicators of plant biomass quality and constitute the most limiting chemical factors to plant productivity in Sub-

Saharan Africa (SSA) farming systems (Smaling et al., 1993; Stoorvogel et al., 1993). Since the early 1990s' numerous investigators have assessed soil nutrient budgets for agricultural ecosystems using mainly a universal mass balance principle, e.g., Cooke, (1958, 1986); Follett et al., (1987); Smaling et al., (1993); Stoorvogel et al., (1993); Tan et al., (2005); Smaling et al., (2012). A conceptual model for soil nutrient budgeting based on a universal mass balance principle was developed (Smaling et al., 1993) to identify components and parameters that are used to characterize both nutrient inputs and outputs (Table 1.3), and the net difference between inputs and outputs of nutrients integrated over a certain area and time to determine the net soil nutrient budget (NSNB). The NSNB depends on the difference between inputs and outputs and may also vary with crop production systems. Agricultural practices with high external inputs likely result in a positive NSNB and may lead to environmental problems by leaching or runoff. On the other hand, agricultural practices with low external inputs likely result in a negative NSNB; nutrient depletion. A balance is only achieved when the nutrient inputs compensate the outputs. Estimates of parameters for different components in Table 1.3 are derived from either pedogenic transfer functions or empirical models (Bouma and Van Lanen, 1987; Van Diepen et al., 1991; Smaling et al., 1993; Stoorvogel et al., 1993). A continental nutrient balance study for Africa reveals that net flows were negative, i.e., 22 kg N, 2.5 kg P, and 15 kg are lost annually per hectare from 1982 to 1984 (Stoorvogel and Smaling, 1990; Stoorvogel et al., 1993). This study brought to prominence the large and continous soil nutrient depletion facing many smallholder African farmers to date and its likely constraint on future food production. The study triggered substantial debate on soil fertility management in SSA and the role of fertilizers, culminating in the involvement of many donor agencies, as well as political commitments on fertilizer use at the Africa Fertilizer Summit in Abuja in 2006 (Sanginga and Woomer, 2009). Nevertheless, these studies have their limitations because they are generalizations at national and continental scales, and only few studies have focused on specific crop production. There are still gaps in the assessment of existing adverse impacts of the negative nutrient budget of a range of crops and cropping systems for soil-specific situations under different levels of nutrient inputs in sub-Saharan Africa (Tan et al., 2005; Smaling et al., 2012). Spatially explicit methods are required to quantify the different nutrient balance parameter and improved the accuracy of the nutrient balance.

Table 1.3: Components of input and output for soil nutrient budgeting

Input	output
IN 1: Mineral fertilizer	OUT 1: Crop products
IN 2: Organic fertilizer	OUT 2: Crop residues
IN 3: Deposition	OUT 3: Leaching
IN 4: N-fixation	OUT 4: Gaseous loss
IN 5: Sedimentation	OUT 5: Soil erosion

1.3 Land degradation in the Lake Victoria basin

The Lake Victoria basin (LVB) is a prime example of an area experiencing high land degradation, climate change, population pressure on land resources and increasing food and energy demand, which together pose a threat to the basin ecosystem productivity. The major concerns are: soil degradation (i.e. organic matter changes and nutrient mining), increased soil erosion from the farms and increased sediment and nutrient loads in the river systems, low and stagnant crop yields, biodiversity loss, and eutrophication of the Lake itself (World Bank, 1996; Scheren et al., 2000; Machiwa, 2003; Lubovich, 2009; Musahara and Rao, 2009). The lake is the world's second largest freshwater body (68,800 km^2) with a catchment area of 250,000 km^2 (UNEP, 2008). The LVB is characterized by the diversity of its landscape and biology. The lake is the defining feature of the regional ecology, the primary source of livelihood security for the basin's human population, and an important source of revenues and economic growth for the nations (Uganda, Kenya, Tanzania, Burundi, and Rwanda) of the catchment basin. Subsistence agriculture, livestock and fishing form a basis for the livelihood of the majority (over 90%) of the human population (30 million people) in the Lake catchment (Isabirye, 2005; Kagera Basin Monograph, 2008).

Land-use change over the years is one of the factors positively correlating with deterioration in lake water quality (Bolstad and Swank, 1997). The Basin has a long history of human-induced LUCC dating back to the colonial era (Olson, 1994) and has been transformed from a relatively pristine system of forests, woodlands and savannas to one of land systems altered by deforestation, biomass burning, livestock keeping, human settlement, urban agriculture and agricultural use. Rapid rural appraisal reports and anecdotal evidence show that this region experienced large-scale land clearing and conversion to agriculture over the last 50 years (Magunda et al., 1998; Isabirye et. al., 2001; Kagera Basin Monograph, 2008) but there have been no long-term studies quantifying LUCC (Isabirye et. al., 2001; Kagera Basin Monograph, 2008). The consequences of LUCC on water and nutrient cycles are largely unstudied. Agricultural expansion and intensive tillage has led to deteriorating soils, leading to increased soil erodibility, accelerated erosion rates and contamination of water. The ensuing land degradation has had a knock down effect on food production and therefore on human livelihoods around the lake (Musahara and Rao, 2009). Stagnation in agricultural productivity rates, combined with limited options for expansion and an increasing population, resulted in declining food production per capita (Ansoms, et al., 2008). Land fragmentation is a serious problem as land pressure increases, resulting in intensive cultivation and land degradation (Ansoms, et al., 2008). The effect of land fragmentation and declining soil quality is manifested through the observed shift from perennial to annual crops, an effort by smallholder farmers to buffer their food security deficits

(Clay et al., 1998; Baijukya, 2004). In addition, cropping areas extend down to streams and lakes edges, eliminating riparian buffering vegetation. The poor land management has resulted in large areas being subjected to severe soil erosion (Scheren et al., 2000; Kagera Basin Monograph, 2008). These developments have led to increased levels of non-point sources pollution loading into Lake Victoria causing of eutrophication (FAO, 1991; World Bank, 1996; Meertens et al., 1995; Meertens and Lupeja, 1996; DANIDA, 1998; Machiwa, 2003; Lubovich, 2009).

Other causes of eutrophication are high export of nutrients that enter streams and lake mainly through atmospheric deposition, runoff and storm water and sediments from urban settlements (Scheren et al., 2000; Isabirye, 2005). The problem associated with sediment transport is that it is a carrier for nutrients, heavy metals and pesticides that adversely affect water quality. Urban pollutants (point source) include untreated municipal sewage, runoff, storm water, agricultural waste, industrial solid waste and animal waste. Industrial and domestic sources of pollution are generally well understood and quantification is relatively straightforward. These sources are localized near urban centres in the immediate vicinity of the lake (Scheren et al., 2000). Pollution from rural areas is thought to be caused by large LUCC to agriculture and increased soil erosion from the farms and increased sediment and nutrient loads in the river systems (Calamari et al., 1994; Musahara and Rao, 2009. How long-term changes in land use and land cover alter soil erosion dynamics in the basin is still unstudied.

Both recent and paleo-limnological investigations have been used to identify sources of nutrients causing eutrophication in Lake Victoria. Most paleo-limnological studies have concluded that sediments from agricultural fields are major sources of P that end up in the lake. Increases in phytoplankton production developed from the 1930s' onwards, which parallels human-population growth and increased landscape cultivation in the Lake Victoria basin (Verschuren et al., 2002; Thomas et al., 1999). A study on lake sediments by Hecky, et al (2000) confirms a 2-3 fold increase in P loading, higher than the 10 $\mu g\ l^{-1}$ observed in an offshore station by Talling, (1965) over the past 50 years with changes in the lake ecosystem beginning even earlier in the century. This agrees with reports of Lehman and Branstrator (1994) that phosphate concentration in Lake Victoria has more than doubled. Higher values of NO_3–N were observed frequently in the lake, particularly at mouths of rivers. Algae concentrations are three to five times higher today than in the 1960s (GIWA 2006). The pattern of phosphorus and nitrogen loading into the lake corresponds with the observed increase in chlorophyll-a concentration and algal blooms during rainy seasons, which can only partly be linked to nutrient upsurge from sediment export and run-off (Ochumba and Kibaara, 1989; Scheren et al., 2000; Mwanjaliwa, 2005).

These observations support the suggestion that phosphorus and nitrogen are the major drivers of the Lake Victoria eutrophication process (Bootsma and Hecky, 1993). Sedimentation rate increased from a previous 57 g m^{-2} yr^{-1} to 90 g m^{-2} yr^{-1} after 1960. The higher turbidity at mouths of rivers is mainly due to suspended sediments. This is a consequence of increased soil erosion (World Bank, 1996; Odada, et al, 2001; Ojok, 2002). Past surveys on eutrophication and water pollution loads from rivers in the northern half of the Lake Victoria Basin (Kisumu) observed that the sediment load from the rivers increased by 7.5 times during 16 years (from 1986 – 2001). Rivers passing through forested lands were less enriched by nutrient as compared to those crossing agricultural lands (Chabeda, 1983; Odada et al. 2001). Studies by Lake Victoria Environmental Management Project (LVEMP) carried out between 2000 and 2005, estimated that 4,905 kilo ton yr^{-1} of suspended sediment load are deposited into Lake Victoria, of which Kagera river, the largest catchment contributes 26.1 % (Myanza et al, 2005). The Kagera river catchment is subject to large LUCC, high-intensity storms and rapid runoff from steep terrain and contributes the largest water inflow into the lake (Kagera Basin Monograph, 2008).

These studies clearly point out the role played by land-use systems and their management in deteriorating of water quality in Lake Victoria and its tributaries. Long-term sustainability of soil and water resources in the Lake Victoria basin (LVB) therefore hinges upon improved understanding of the connectivity of the following biophysical processes; LUCC, soil quality changes, soil erosion and nutrient flow dynamics. The biophysical factors are conjoined and reverberate together as part of a cascading ecosystem framework, potentially contributing to soil degradation and eutrophication in the LVB. The understanding of these processes is, however, fragmented and dispersed, and analyses that link land and water resource degradation in the Lake Victoria basin are scarce.

1.4 Research problem

Whereas the impact of eutrophication on the declining lake ecosystem functions and productivity has been quantified and well documented (e.g, Hecky et al., 2000; Scheren et al., 2000), the source of nutrient loading remains a controversial issue. A study by Scheren et al., (2000) on a mass balance assessment of nutrient loading into Lake Victoria revealed that nutrient loading into the lake is mainly associated with atmospheric deposition and rural soil degradation. These two together account for about 90% of the phosphorus and 94% of the nitrogen import into the lake. Rural soil degradation due to agricultural land uses directly contributes 55 % phosphorus and 22 % nitrogen export into the lake. The study revealed that the industrial pollution of streams is still insignificant to the role of nutrient loading into the rivers and the lake, because of the weak industrialization development. The water courses and their water quality are mainly influenced by rural agricultural land use activities (Musahara and Rao, 2009). Large expanses of rural smallholder farms are therefore implicated in actuating soil degradation and eutrophication in Lake Victoria. Land-use change over the years is one of the factors positively correlating with deterioration in lake water quality (Bolstad and Swank, 1997). Despite earlier studies (e.g. Tukahirwa, 1996; Bagoora, 1998; Mulebeke, 2004; Majaliwa, 2004), what is still unclear is the nature, extent and trend of LUCC, rural soil degradation and potential sources of pollution loading into the river/ lake system. Are they coming from the immediate periphery of the lake or from the upstream of the watershed? Available studies have been conducted downstream near the Lake Victoria basin and focused on individual processes such as plot and micro catchment soil erosion and water quality. These studies cannot be aggregated to provide sufficient understanding of the dynamic relationship between landscape degradation and eutrophication in Lake Victoria. Much of the uncertainty stems from lack of additional studies on long-term LUCC impacts on soil quality changes, hydrological fluxes and soil erosion dynamics in rural catchments especially in upstream areas of large river systems such as the Kagera river despite the fact that large LUCC occurred here (Isabirye, 2005; Kagera Basin Monograph, 2008; Musahara and Rao, 2009). To advance knowledge on this front, transformations in LUCC, soil quality changes, and erosion dynamics in the upstream catchment of the Kagera river Basin were studied in this thesis. Figure 1 shows the research process. Following the Drivers-Pressures-State-Impact-Response (DPSIR) framework (Smeets and Weterings, 1999, Smaling and Dixon, 2006), land and water resource degradation process in the Lake Victoria basin can be categorized as a connection between land degradation drivers and pressures exerted on the land by human activities and natural phenomena, the consequent changes in quality of the resource, and the importance of responding to these changes as society attempts to release the pressure or to rehabilitate land which has been degraded. The interchanges among these form a continuous feed-back mechanism that can be monitored and used for the assessment of land quality and sustainable environmental management.

1.4.1 Research questions

The overall question is: how do changes in land use and land cover alter soil fertility and soil erosion dynamics? This question leads to the following specific questions and hypothesis addressing key elements of this study:

1) What are the land use changes in the last 100 years?
2) What controls variation in carbon and nutrient stocks, and how are these variations modified by LUCC?
3) What are the current erosion patterns and catchment soil loss rates?

4) What is the simulated change in sediment dynamics from 1974 until now as a result of land use change?
5) What is the nutrient balance of the different farming systems and what role does erosion play in this balance?
6) What are the implications for sustainable land management?

The working hypothesis of this study is that land use and land cover change has a significant effect on soil erosion, carbon stocks, soil fertility and pollution loading.

1.4.2 Objectives of the study

1.4.2.1 Overall objective

To contribute to understanding of how land use and land cover change affect hydrology, sediment fluxes and soil fertility as a scientific basis for evaluation of the dynamics of sediment/ nutrient loading and eutrophication in Lake Victoria.

1.4.2.2 Specific Objectives

The objectives of the study are:

1. To characterize and quantify historical land use and land cover changes (1901 – 2010) in Kagera river watershed, the upper part of LVB catchment
2. To estimate historical and contemporary changes in Carbon stocks associated with land use changes
3. To estimate spatial and temporal patterns of soil loss associated with land use changes
4. To assess changes soil nutrient stocks associated with cropping systems and landscape positions

1.5 Outline of the thesis

The thesis examines the influence of long term land use and land cover changes (LUCC) on soil organic carbon (SOC), erosion dynamics and soil fertility in the upper reaches of the Kagera river basin. The thesis write-up focused on six chapters. Chapters 2 to 5 are developed as separate articles following the objectives of the thesis research.

Chapter 1 is a general introduction on research issues in the Lake Victoria Basin.

Chapter 2 assesses land cover changes over 100 years since 1901 and identifies hot spots and potential impacts on spatial environmental quality. This chapter is already published as; *Wasige, E.J., Groen, T.A., Smaling, E.A.M., Jetten, V., 2013.* Monitoring basin-scale land cover changes in Kagera

Basin of Lake Victoria using ancillary data and remote sensing. International Journal of Applied Earth Observation and Geoinformation 21, 32 – 42

Chapter 3 reports on studying the effect of LUCC on soil organic stocks in the Upstream Catchment of Lake Victoria Basin, South-West Rwanda. Submitted as: *Wasige, E.J., Groen, T.A., Rwamukwaya, M.B., Tumwesigye, W., Smaling, E.M.A., Jetten V.,* Contemporary Land Use/ Land Cover Types Determine Soil Organic Stocks in the Upstream Catchment of Nile Basin, South-West Rwanda. Journal of Nutrient Cycling in Agroecosystems

Chapter 4 presents a spatially explicitly distributed hydrological model that was developed. This model requires relatively little data and has a better capability of predicting and evaluating the effects land use change and conservation management on soil erosion from field to catchment scales compared to current spatially distributed erosion models. Submitted as: *Wasige, E.J., Jetten, V., Groen, T., Smaling, E.M.A.,* A New Spatially Explicit Hydrological Model for Erosion Modeling: Bridging the gap between farming systems to catchment soil loss in the highlands of Rwanda. Journal of Hydrology and Earth System Sciences (HESS)

Chapter 5 reports on the effect of permanent cropping and low nutrient input systems on soil fertility and nutrient balances in South-West Rwanda. This chapter explores key challenges and opportunities for soil fertility management for sustainable agricultural production and food security. Submitted as: *Wasige, E.J., Groen, T., Smaling, E.M.A., Jetten, V.,* Soil fertility and nutrient balances of low input land use systems of South-West Rwanda, Upstream of Lake Victoria Basin. Journal of Agricultural ecosystem and environment

Chapter 6 is the synthesis on the major findings of the thesis, discusses hypothesis, nature, extent and trends of LUCC, outlines strategies for replenishing SOC, soil fertility and restoring the nutrient balance in farming systems. Recommends are made on new areas for research and policy development to support decision making for sustainable management of land resources. Besides, the model from chapter 4 was used to assess how the catchment system will respond to soil conservation interventions in the future.

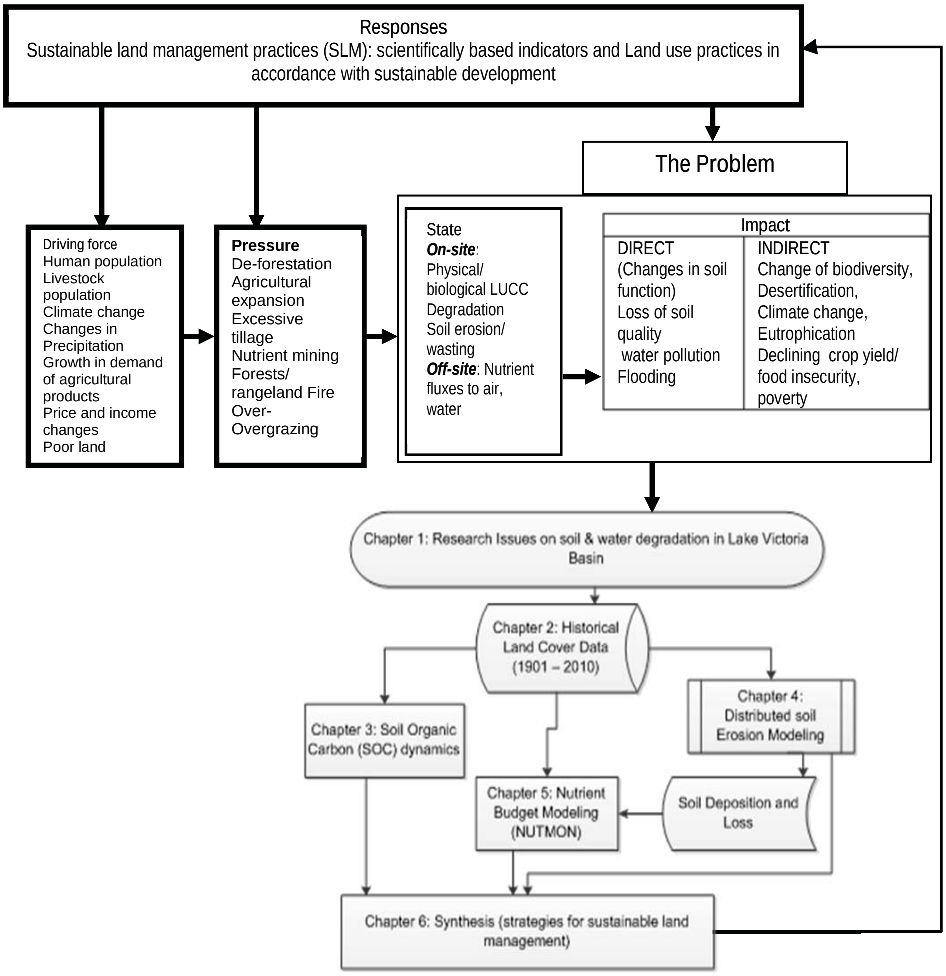

Figure 1.1: DPSIR frame work indicators highlighting research activities

Chapter 2

Monitoring basin-scale land cover changes in Kagera Basin of Lake Victoria using ancillary data and remote sensing

Abstract

The Kagera basin is a high value ecosystem in the Lake Victoria watershed because of the hydrological and food services it provides. The basin has faced large scale human induced land use and land cover changes (LUCC), but quantitative data is to date lacking. A combination of ancillary data and satellite imagery were interpreted to construct LUCC dynamics for the last century. This study is an initial step towards assessing the impact of LUCC on sustainable agriculture and water quality in the watershed. The results show that large trends of LUCC have rapidly occurred over the last 100 years. The most dominant LUCC processes were gains in farmland areas (not detectable in 1901 to 60% in 2010) and a net reduction in dense forest (7% to 2.6%), woodlands (51% to 6.9%) and savannas (35% to 19.6%) between 1901 and 2010. Forest degradation rapidly occurred during 1974 and 1995 but the forest re-grew between 1995 and 2010 due to forest conservation efforts. Afforestation efforts have resulted in plantation forest increases between 1995 and 2010. The rates of LUCC observed are higher than those reported in Sub Saharan Africa (SSA) and other parts of the world. This is one of the few studies in SSA at a basin scale that combines multi-source spatio-temporal data on land cover to enable long-term quantification of land cover changes. In the discussion we address future research needs for the area based on the results of this study. These research needs include quantifying the impacts of land cover change on nutrient and sediment dynamics, soil organic carbon stocks, and changes in biodiversity.

Keywords: *Historical analysis*, *data integration, Land use/land cover changes; land degradation, GIS/Remote Sensing, eutrophication, Lake Victoria Basin*

2.1 Introduction

Land use and land cover change (LUCC) is a well-recognized agent of ecological change and a prominent interface between human activities and global environmental change. It is a fundamental process that impacts on, and links many parts of the environment by altering complex biophysical processes (such as energy and mass flux) at global, regional and local scales (Houghton, 2003; Wu, et al. 2003). The interactions of people and the environment mostly play out at the land surface (Lambin and Geist, 2006). This might take the form of conversion of natural landscapes for anthropogenic use or by changing management practices in human-dominated landscapes (Foley, et al., 2005). Significant bodies of research link LUCC to approximately 20 % of the global CO_2 emissions to the atmosphere (Houghton, 1995, 1999, 2005; Batjes, 2004; IPCC, 2007; Van der Werf, et al. 2009) and the greater part of these emissions are coming from the tropics (Rhoades, et al. 2000; IPCC, 2007; Verburg, et al. 2011). About half of the ice-free land surface has been converted or substantially modified by human activities over the last 10,000 years (Lambin, et al. 2003). It is estimated that undisturbed (or wilderness) areas represent 46% of the earth’s land surface (Mittermeier, 2003). Forests covered about 50% of the earth’s land area 8000 years ago (Ball, 2001) as opposed to 31% today (FAO, 2011). Global extents of croplands, pastures, plantations, and urban areas have expanded in recent decades. Wide-ranging LUCC are driven by the need to provide food, fibre, water, and shelter to more than seven billion people (Foley, et al., 2005; Lal, 2009; Smaling, et al. 2012). Such changes have enabled humans to appropriate an increasing share of the planet's resources, but they also potentially undermine the capacity of ecosystems to sustain soil quality, food production, maintain freshwater and forest resources, regulate climate and air quality (Foley, et al., 2005). This trend of LUCC has attracted attention because of the potential effects it has on the biogeochemistry, hydrology, food security, climate and socioeconomic systems (IPCC, 2001, 2007; Houghton, 2003; Veldkamp and Verburg 2004; Lal, 2009; Smaling, et al. 2012) of locations. This may considerably affect the land capacity to sustain biological productivity and to maintain environmental quality and long-term sustainability of socioeconomic systems (Vitousek, et al. 1997).

Understanding the role of land use in global environmental change and the long-term human-environment nexus requires historical reconstruction of past land use and land cover conversions. The information on land use changes in most developing countries is usually missing, out dated or inconsistent (Di Gregorio and Jansen, 1998; Brink and Eva, 2009). The launch of the Landsat imagery platform in 1972, followed by others such as SPOT and ASTER, has provided satellite remote sensing (RS) capacity to detect LUCC over the past 40 years at most. For the period before satellite remote sensing, ancillary data such as, historical maps, aerial photographs and topographic maps can be used. Several studies have attempted to reconstruct spatially explicit historical land use changes by combining; RS and ancillary data (Muller, et al. 1999, Salami, et al. 1999, Reid, et al. 2000; Petit and Lambin, 2001a; Petit and Lambin, 2002; Hepcan, et al. 2010), RS data and historical inventory data (Esser, et al. 1994; Klein Goldewijk and Batjes, 1997; Ramankutty and Foley, 1999; Ramankutty, et al. 2002; Lambin, et al. 2003), or by using land use change models (Zuidema, et al. 1994; Veldkamp and Fresco, 1996; Brown, et al. 2007; Pontius Jr, et al. 2008). Among these studies, Petit and Lambin, (2001a) compared the application of land use change models and data integration (a combination of RS data and ancillary data) of a series of historical land cover maps for reconstruction of historical LUCC. They concluded that there was agreement, with differences lower than 6% observed between modelling and data integration approaches. For our study, we chose to use the data integration approach to construct long-term LUCC because of availability of historical maps constructed from ancillary and RS data.

It should be noted that the terms land use and land cover are not identical and environmental studies draw attention to their differences. "'Land cover' refers to the biophysical state of the earth's surface and immediate subsurface" (Turner et al. 1995), on the other hand, "'Land use' denotes the human employment of land" (Turner and Meyer 1994). FAO (2000) states that land use is characterized by the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it. The fundamental definition of the land use establishes a direct link between land cover and the actions of people in their environment that bring land use changes. The combined use of land cover and land use data allows detection of where certain changes occur, what type of change, as well as how the land is changing (Jansen and Di Gregorio, 2002). In this study we will quantify changes in land cover, but with additional knowledge of the processes on the ground, inferences will be made about the use of the land in the discussion. For succinctness the term LUCC will be used throughout the document to indicate the changes detected and the underlying uses associated with them.

Sub Saharan Africa has experienced long-term LUCC but only a few and patchy studies exist on long-term characterization of LUCC (Biggs and Scholes, 2002). Examples include; Tappan, et al. (2000) in Senegal, Bewket, (2002) in Ethiopia and Biggs and Scholes, 2002 for South Africa. At basin scale, most of the major African river and lake basins are undergoing large scale LUCC due to agricultural expansion, mainly to grow food for the ever expanding population but there are few studies documenting long-term LUCC in these basins (Rangeley, et al. 1994). Conventionally, data on land cover are available through the Food and Agricultural Organization (FAO) of the

United Nations, which collates statistics from the countries themselves, harmonizes and rationalizes them to form global data bases on forests (Forest Resource Assessment – FRA) and on agricultural production (FAOSTAT). While these databases are valuable sources of information, they inevitably suffer from a lack of consistency, unknown errors and completeness both in time and in geographic coverage, often relying as they do on the completeness of the national data sources which in some countries are incomplete or out of date. The methods for collecting the data, the actual information collected and aggregation it undergoes may vary markedly between countries, making harmonization and rationalization difficult (Eva, et al., 2006; Brink and Eva, 2009).

The Kagera basin is a global hotspot of large and long-term human induced LUCC transformations from a relatively pristine system of tropical forests, woodlands and savannas to one of land systems altered by deforestation towards agricultural use. In the last 50 years, the impacts of land use change have increased from significant to threatening proportions (World Bank, 1996; Isabirye, 2005; Kagera Basin Monograph, 2008; Musahara and Rao, 2009). The major environmental problems related to LUCC in the basin include; soil degradation, siltation from the erosion of deforested landscapes, eutrophication, desertification, biodiversity loss and local climate change. Despite the increasing recognition that the Kagera basin has faced large LUCC, the extent, nature, magnitude and rates of LUCC in this region are still unclear. Lack of LUCC data is well recognized as a critical gap in the knowledge of soil and water degradation, biodiversity loss, soil erosion and eutrophication in the lake Victoria Basin. The goal of this study was to detect land use changes and to characterize the processes of land cover change in terms of their spatio-temporal pattern over a period between pre-1901 to 2010 for the Kagera Basin. Three questions were addressed in this study; (1) what is the magnitude of land cover change—*how much*? (2) where are the areas most affected by land-cover change—*where? (Spatial analysis)*; (3) At what rate did the land-cover change progress and when did it start— *when? (Temporal analysis)*. This study is an initial step towards assessing the impact of LUCC on sustainable agriculture and water quality in the Lake Victoria catchment and is performed in a part of the world that has high LUCC dynamics. The ultimate goal is to identify the most important changes and with that prioritize research needs related to the impacts of these LUCC dynamics.

2.2 Materials and methods

2.2.1 Location and description of the study area

The study site was Kagera transboundary river basin of the Lake Victoria catchment (Figure 1.1). The river basin lies between 0^0 45’ and 3^0 35’ South latitude and $29^0$15’ and $30^0$51’ East longitude and falls under the four countries of; Burundi (22%), Rwanda (33%), Tanzania (35%) and Uganda (10%). The basin occupies a strategic position in Africa providing a major catchment for the world’s second largest lake and up to 10 % of the water of the downstream Nile Basin. The basin straddles an area of about 60,000 km^2 and forms the upper part (75 %) of the Lake Victoria basin, making the Kagera River the largest water inflow into Lake Victoria with an estimated 7.5 km^3 per year. It is believed to contribute the largest sediment and pollution loading into Lake Victoria (Kagera Basin Monograph, 2008; Musahara and Rao, 2009). The basin is characterized by three climatic zones (humid, sub-humid and semi-arid) and three different farming systems adapted to these different rainfall and topographic conditions. The agricultural systems include an agro-pastoral system in the savannas and semi-arid areas downstream and a banana - coffee - mixed annual cropping system in the humid upstream areas on the other side of the spectrum. In between the two farming systems, there is a mixed crop-livestock system in the sub-humid area. Vegetation types range from a complex of forests and woodlands, savannas, shrublands to aquatic vegetation in wetlands. The largest primary forest area, Nyungwe forest, occurs in the upstream part of the river basin which has a humid climate and is also considered a “water tower” of the Lake Victoria basin. Mean annual temperature is about 15^0 - 18^0C upstream and 21^0 - 30^0C downstream. The rain pattern is bi-modal, with long rains occurring during September to January, and shorter rains in March to June. Annual precipitation ranges from over 2000 mm upstream to 800 mm downstream with high variation. Rain falls mainly during storms that produce large amounts of runoff that consequently can become problematic by causing floods or acting as driver of soil erosion (Kagera Basin Monograph, 2008; Musahara and Rao, 2009).

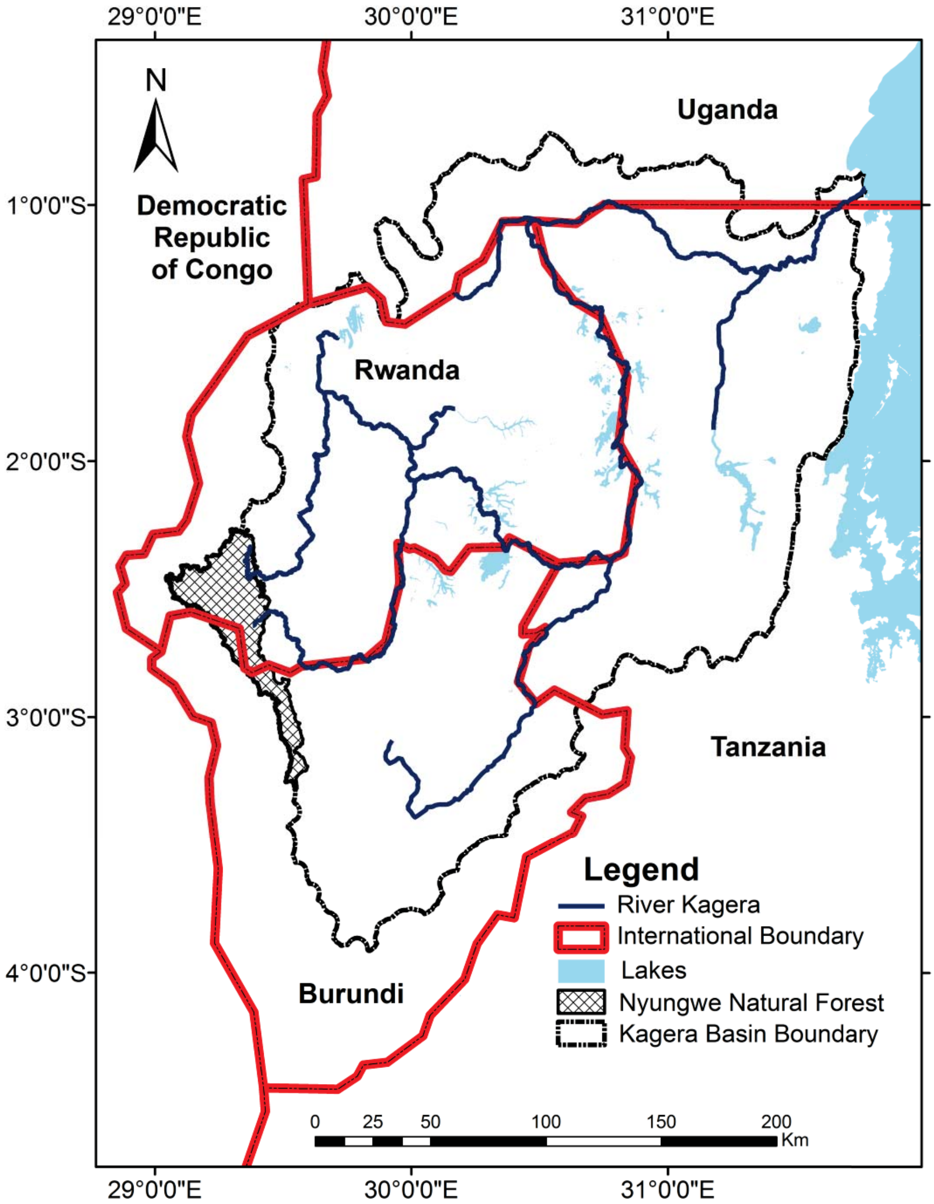

Figure 2.1: Location of Kagera Basin in the transboundary countries

2.2.2 Land cover classification

This study was based on data integration of a time series of ancillary data (1901 historical thematic map, topographic sheets of 1974 and 1988, interviews, and literature review) and satellite RS data to assess trends in long-term LUCC. Historical thematic maps of pre-1901 developed by Prioul and Sirven (1980) and the Lands and survey department of Uganda *(*Atlas of Uganda, 1967) were scanned, geo-referenced and digitized to construct the land cover map of pre-1901. The pre-1901 land cover map was consistent with the geo-referenced 1974 topographic sheets in terms of landscape features like lakes and river streams. Topographic sheets (1:50.000) constructed from aerial photographs of 1974 were scanned, geo-referenced and digitized to create a land use/land cover map of 1974. While considering image availability and quality, a time series of predominantly cloud free Landsat scenes of the years 1985, 1995 and 2010 was chosen (Table 2.1) making it feasible to map LUCC at a medium scale, appropriate for detecting large-area changes and with minimal bias from seasonality. We also downloaded and used processed Landsat ETM+ images that have the SLC-off data gaps/ stripping problem resolved by United States Geological Survey (USGS). Efforts were made to obtain the images from the same year and where there was difficulty in image availability for a particular targeted year, images from the previous year or the next year (minus one or plus one year) were considered. At present, there is no standard land use and land cover classification system for the Kagera Basin, although the FAO\UNEP land cover classification system (LCCS) exists, which is widely accepted as a standard for LUCC studies. However, for the purpose of this study, these classes are still too broad, and therefore the USGS land use/land cover and IGBP DISCover classification system (Anderson, et al., 1976; Belward, 1996) was adapted and modified to include classes that are applicable to the study area and general enough so that they can be readily identified on Landsat imagery (see Table 2.2). For compatibility with the LCCS system, the key to this system is included in Table 2.2. The acquired Landsat images were geo-referenced to the Rwandan projection (local Transverse Mercator coordinate system) of the 1:50.000 topographic maps, radiometrically normalized (Price, 1987; Markham and Barker, 1988) and rectified to each other. Further corrections to the images included atmospheric normalization (Richter, 1997) using dark pixel subtraction (Elvidge et al., 1995) to remove haze (Du et al. 2002). Geo-coded images were then overlaid with vector data (roads, rivers and lakes) where a perfect match was observed and then sub-setted to the extents of the study area. Supervised classification procedures using Erdas2010 software were implemented to classify the Landsat images into the established land cover classes using the Maximum Likelihood Classification (MLC) algorithm. Isolated pixels of a class were eliminated and replaced with values based on their surroundings using a 3x3 majority filter. Topographic maps of 1988 with a 1:50.000 scale were used to validate the 1985 Landsat classification map with the assumption that there were no major LUCC between the two periods. The final classification and interpretation of the 2010 land cover map was conducted based on field observation data and 2008 aerial photographs (available at Rwanda's National Land Centre) as the reference information for validation and accuracy assessment. We used a handheld GPS to locate land cover classes with an accuracy of between 1 and 5 m during the field observations. A total of 162 GPS points covering all land cover types in the basin were collected. Clouds and cloud shadows were masked from the images by manual interpretation and digitizing. The classification accuracy of each type of image was evaluated and a confusion matrix was produced, from which the overall accuracy and kappa statistics of agreement for each land cover category were computed (Yang and Lo, 2011). It was difficult to carry out accuracy assessment for the 1901, 1974, 1995 land use/cover maps because of the lack of aerial photographic coverage or any historical reference data for this period. We relied on the quality of the original status of the datasets retrieved. However, we assumed that accuracy assessment for the land use/cover maps of 1986 and 2010 should be sufficient to shed light on the overall accuracy of the land use/cover mapping procedures adopted in this study. To establish the LUCC that occurred during the 1901–2010 period, a post-classification change detection analysis of the land cover maps was performed using ArcGIS10 overlay operations. For all change maps 'from-to' LUCC matrices were extracted (Table S2.3 – S2.6 in appendix 2.1: supplementary materials).

Table 2.1: Attributes of the Landsat TM/ETM+ imagery used in the study

Acquisition	Sensor	Spatial resolution (m)	Path/row
1984-06-20	TM 5	30	172/61
1995-01-10	TM 5	30	172/61
2010-08-15	7 ETM+	30	172/61
1984-06-20	TM 5	30	172/62
1995-01-10	TM 5	30	172/62
2009-06-25	7 ETM+	30	172/62
1984-07-19	TM 5	30	173/61
1995-01-17	TM 5	30	173/61
2010-01-26	7 ETM+	30	173/61
1986-07-19	TM 5	30	173/62
1995-01-17	TM 5	30	173/62
2010-08-22	7 ETM+	30	173/62

Table 2.2: Description of land use/land covers classes and definitions

IGBP DISCover classes	Definitions	Equivalent of UN land cover classification system (LCCS) classes
Dense Forest	Native forest before it's cleared for other land uses. Mature vegetation >5m; >60% ground surface covered by trees; almost all trees remain green all year; canopy is never without green foliage	Forest
Degraded Forest	Primary forest degraded by selective logging and fire. Includes coniferous, deciduous and mixed forests. Mature vegetation >5m; > 40 % to <60% ground surface covered by trees	Degraded forest
Woodlands	Closed shrublands, open shrublands and Woody Savannas. Mature vegetation >5m for closed shrublands, open shrublands and <5m Woody Savannas; tree canopy cover >10% and <60%; bush and shrub coverage >40%	Deciduous shrubland with sparse trees
Savannas	Shrubs <2m; shrub canopy coverage <40%; remaining cover either bare or annual herbaceous/ grasses; shrubs either evergreen or deciduous	Open grassland with sparse shrubs
Tea	The tea-plant, in the natural state, grows into a small or medium-sized tree, but it is pruned and trained to knee height (< 0.5m) form a many-branched low bush mart. >70% ground surface covered of tea plants	Tree crops
Plantation Forest	Manmade forest developed by planting trees; >40% ground surface covered by trees	Plantation forest
Bamboo	Bamboo belongs to the family of grasses. Most Bamboos are treelike and shrubby grasses with woody stems. You can find Bamboos in tropical and subtropical to mild temperate regions. Bamboo range from stiff reeds about > 1m tall to giants reaching 50 m in height and 30 cm in diameter near the base	Bamboo forest
Water Bodies	All areas of open water, including streams, rivers and reservoirs	Inland waters
Farmlands	Land cultivated for food and fiber; >80% of landscape covered in crop-producing fields and includes cultivated land without crops,	Croplands
Urban and Built-up	Land covered by buildings and other manmade structures	Cities
Permanent Wetlands	Lands with a permanent mixture of water and herbaceous or woody vegetation that cover extensive areas. The vegetation can be present in either salt, brackish or freshwater	Swamp bushland and grassland

2.2.3 Data integration and error control

Potentially, changes can occur due to the use of multiple data sources with variable precisions. We minimized these errors by adopting map generalization approaches where two map properties were modified as follows (Petit and Lambin, 2001a); firstly, thematic consistency was achieved by harmonization of the legends of the land cover maps to be compared. The initial land cover legend was derived from the land cover map of 1901 with 17 land cover categories and the 1974 topographic maps with 18 land cover categories. The classes of both land cover maps were merged into 11 land cover categories based on their similarity in terms of biophysical attributes (table 2.2). These classes were used to generate land cover maps derived from Landsat imagery for the years 1985, 1995, and 2010. Secondly, the spatial level of detail was standardized by considering a minimum mapping unit of approximately 0.56 km^2 consistent with the lower resolution of the 1901 land cover map (1:150000). The 1974, 1985, 1995, and 2010 land cover maps were equalized using this minimum mapping unit (Townshend, et al*.* 2000; Petit and Lambin, 2001a) in an elimination operation, merging polygons smaller than this unit to neighbouring polygons.

2.3 Results

2.3.1 Accuracy assessment

Results of the classification accuracy assessment for the 1985 and 2010 land cover maps are presented in the supplementary materials; Tables S2.1 and S2.2 (see Appendix 2.1). For the 1985 classification, the overall accuracy was 86%, and the Kappa coefficient 0.83. Most of the land cover classes had producer's accuracy and user's accuracy over 80%, implying high accuracy levels of the classification. However, for some land cover categories we had low accuracy and Kappa coefficient values, e.g. the producer's accuracy for dense forest class was 71% although the user's accuracy is higher (86%). The confusion matrix for the 2010 classification was a slight improvement compared to the 1985 classification product. The overall accuracy was 92% and the Kappa coefficient 0.91. In this classification, the reference data and the classified classes were largely in agreement probably because of the use of improved spatial accuracy of the reference dataset collected by field observations, compared to using a validation dataset of topographic maps from 1988 for the 1985 classification. When the classification accuracy assessment is compared between years, a lower producer accuracy for dense forests (71%) in 1985 compared to 2010 (77%), means part of the detected increases in LUCC for this period can also be attributable to the differences in accuracies. Likewise, a low user's accuracy (67%) for plantation forest in 1985 and a higher user's accuracy (79%) in 2010 means that plantation forest was probably overestimated for 1985 and part of the detected decreases in plantation forest are possibly attributable to differences in accuracies between these years.

Table 2.3: Land-cover proportions between 1901 and 2010 in percentage and annual rates of land-cover changes in per cent for the period of 1901–2010

Land cover classes	1901 (ha)	%	1974 (ha)	%	1985 (ha)	%	1995 (ha)	%	2010 (ha)	%	annual rates of land-cover changes in per cent for the period of 1901–2010
Dense Forest	442978	7.4	314458	5.2	157896	2.6	134308	2.2	154114	2.6	-1.1
Savannas	2114750	35.2	1722895	28.7	1418271	23.6	1409982	23.5	1174285	19.6	-0.7
Woodlands	3035921	50.6	1608521	26.8	923798	15.4	435805	7.3	411975	6.9	-1.7
Degraded Forest	0	0.0	6764	0.1	35958	0.6	50089	0.8	24485	0.4	5.8
Tea	0	0.0	4607	0.1	4641	0.1	7939	0.1	8568	0.1	1.9
Plantation Forest	0	0.0	67296	1.1	152688	2.5	200920	3.3	200321	3.3	4.4
Bamboo	14728	0.2	5015	0.1	5015	0.1	5015	0.1	4747	0.1	-0.1
Farmlands	0	0.0	1877181	31.3	2904541	48.4	3350369	55.8	3615932	60.3	2.1
Urban and Built-up	0	0.0	1641	0.0	5569	0.1	13950	0.2	13950	0.2	16.7
Water Bodies	107401	1.8	107401	1.8	107401	1.8	107401	1.8	107401	1.8	0.0
Permanent Wetlands	284266	4.7	284266	4.7	284266	4.7	284266	4.7	284266	4.7	0.0

2.3.2 Land cover

The results of the land cover classification are presented in Table 2.3 and Figure 2.2. Table 2.3 and Figure 2.2 reveal that around 1901, the land cover was dominated by woodlands (51%) and savannas (35%). By 2010, the dominant land cover shifted to farmlands (60%) and savannas (19.6%). The most dominant cover change was expansion of farmlands (not reported in 1901 to 60% in 2010) while most other land uses experienced reductions in cover during the same period. The observed expansion of farmlands followed generally a west to east trend.

The pattern of land cover conversion is shown in Figure 2.3, which also shows a dominant trend of conversions from west to east. Conversion tables can be found in the supplementary materials; Tables S2.3-S2.6 (see Appendix 2.1), but the most striking conversions are mentioned below. At the start of the studied period, expansion of farmlands was (in order of importance) at the expense of Savannas (1.12 Mha), Woodlands (0.61 Mha) and dense forest (0.11 Mha). Savannas remained over the subsequent periods the most dominant land cover type that was converted into farmlands. The rate of savannas-farmland conversion was highest between 1974-1985 (58 kha yr^{-1}), when the overall rate of farmland expansion was highest for the whole period (81 kha yr^{-1}) as well, and then declined. The conversion of dense forests into farmland also peaked during 1974-1985 (8 kha yr^{-1}) but after that dropped sharply, while the rate of conversion of savannas into farmland

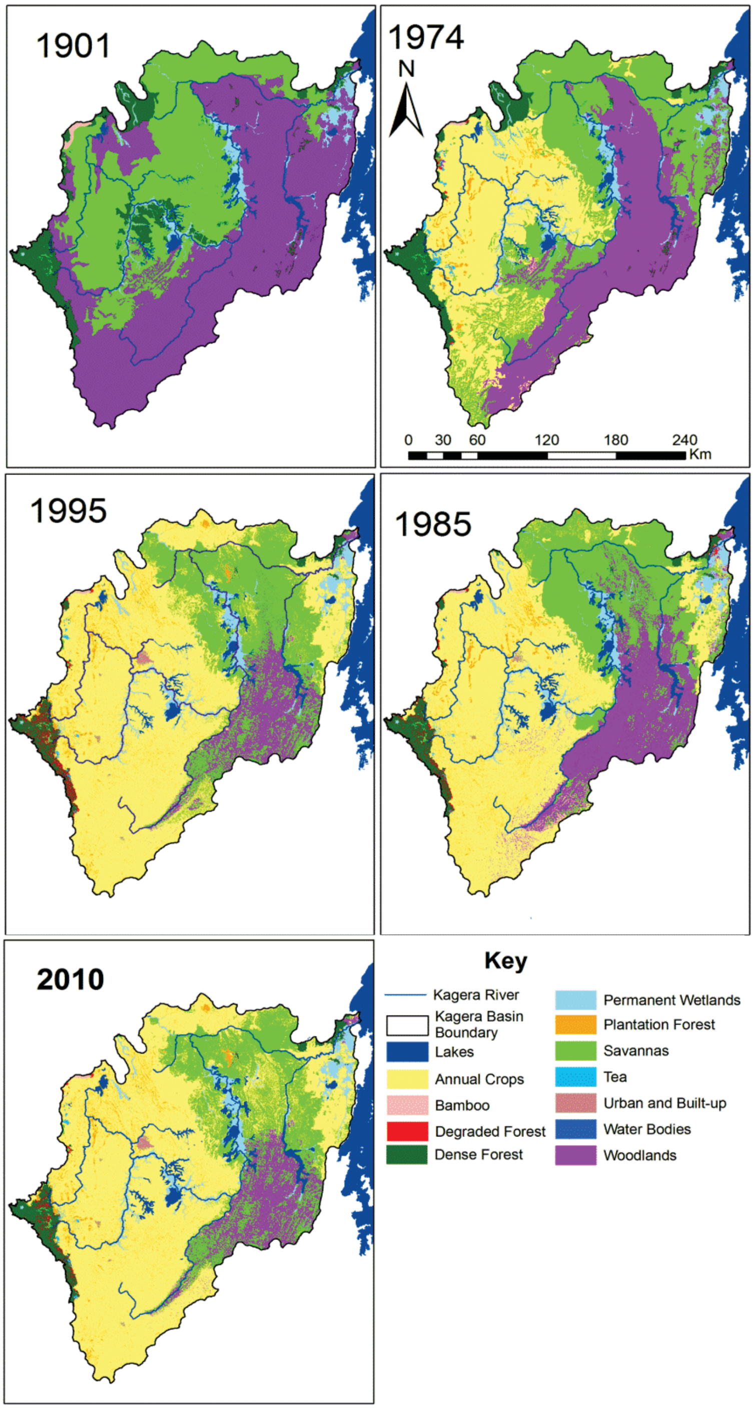

Figure 2.2: Land cover in the Kagera River Basin for the studied years

remained high also for the period 1985-1995 (40 kha yr^{-1}), and only slightly dropped after 1995 (17 kha yr^{-1}). There was also a marked trend in woodland to savanna conversion, which peaked in the period 1985-1995 (41 kha yr^{-1}).

Also overall deforestation was highest between 1974-1985 (11 kha yr^{-1}), and was complemented with deforestation (3 kha yr^{-1}) although the total forest cover in the area stabilized after 1974 mainly due to increases in plantation forest (Figure 2.4). Actually, the area covered by plantation forests became larger than the dense forests after 1984. However, forest degradation continued after that period, especially in the Nyungwe forest.

As mentioned previously, the overall pattern of change moved from the wetter west part of the basin to the drier eastern part of the basin during the investigated period. Most of the conversions towards farmlands occurred in the earlier periods (1901-1984) in the Rwandan and Burundian part of the basin. The conversion of woodlands into savannas mainly occurred in the drier, Tanzanian part of the basin. Especially near the edge of Lake Victoria, a trajectory from woodland to savannas to farmland can be distinguished in the period before 1985. In the Ugandan (*i.e*. northern) part of the Basin, most changes occurred between 1985 and 1995, and concerned the conversion of savannas into farmlands, although in the period 1974-1985 a huge forested area was cleared for farmlands.

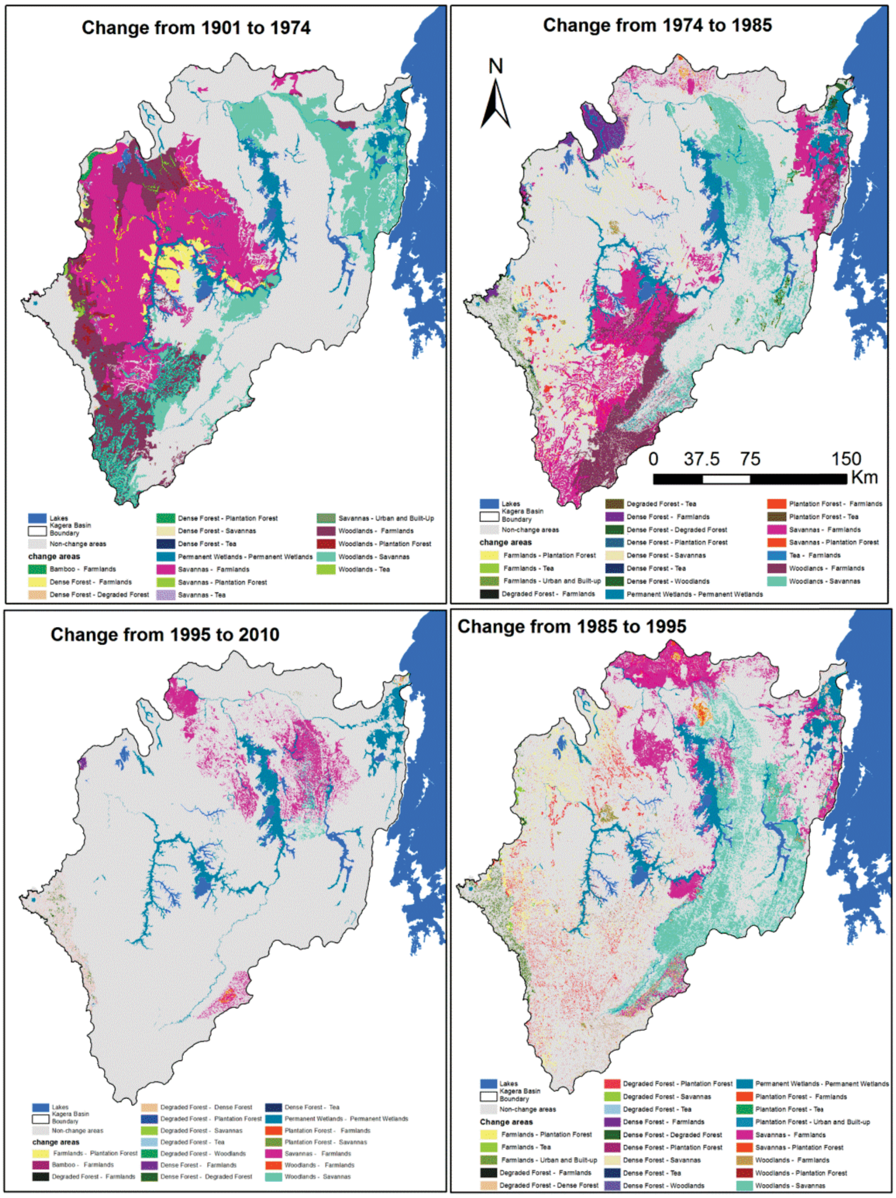

Figure 2.3: Land cover changes per period in the Kagera river basin

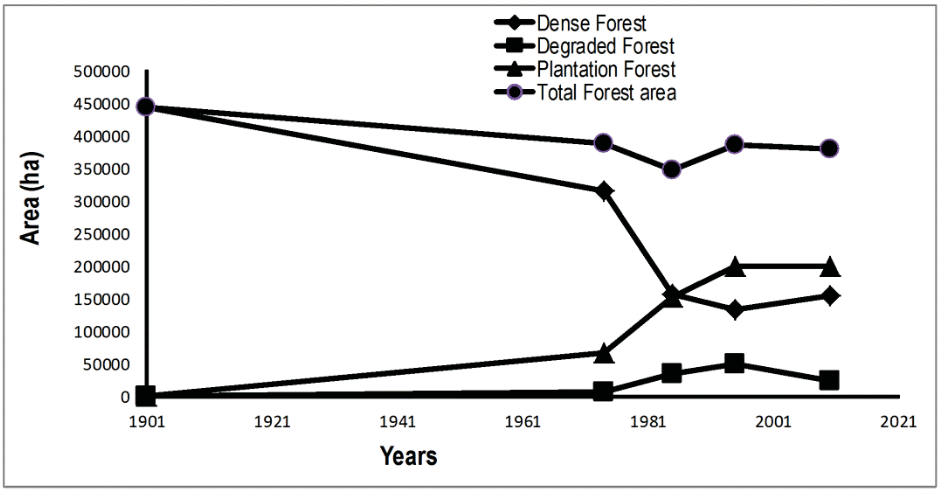

Figure 2.4: Temporal variations in forest cover since 1901

2.4 Discussion

Clearly the results have shown that massive land cover conversions have occurred in the Kagera basin. The overall farmland expansion rate of 0.57% yr^{-1} (calculated over the whole period 1901-2010) is not as high as reported in other studies e.g. 1.6% yr^{-1} for other parts of Uganda (Ebanyat, et al. 2010), 1.91 % yr^{-1} for a region in Zambia (Petit, et al. 2001b) but at certain periods in the studied time frame similar rates were observed (1.56 % yr^{-1} between 1974 and 1985). It is highly likely that these conversions will have had an impact on biodiversity, erosion, nutrient cycling, river water quality and system productivity. However, the magnitude of these impacts will depend heavily on the way the conversions were implemented. For example, conversion of forest into farmland can result in severe erosion when this occurs on the wrong locations, farming techniques are not properly adapted, or when the choice of crop is ill suited. However, when proper measures are taken, erosion might be greatly limited and soil quality might even improve. The likeliness of which of these two scenarios occurred during observed scenarios depends heavily on the socio-economical setting under which the conversions occurred. Therefore, this discussion will focus on two main aspects; (1) the possible drivers of LUCC and how these were set within the socio-economic setting at that time, and (2) the potential implications of the observed LUCC given the drivers, and consequently where research priority should be focused.

2.4.1 Potential Drivers

Trends in population growth, economical developments and land use policy changes are the most likely drivers of the observed large scale LUCC in

Kagera Basin, as has been shown for other parts of Uganda (Ebanyat, et. al. 2010). The current population in the Kagera basin is estimated to be nearly 15 million people, representing about 40% of the total Lake Victoria basin population (World Bank, 2008; Kagera Basin Monograph, 2008). Projections of the future population size suggest it will be increasing even more with average annual population growth rates for the period 1999-2015 estimated at 3.4% (Rwanda), 2.3% (Tanzania) and 3.2% (Uganda) (Kagera Basin Monograph, 2008). The mean estimated population density was 248 persons km^{-2} in June 2007, which is more than 8 times the 28 persons km^{-2} average for SSA. This population density is not equally shared among the Kagera Basin countries; the population density is 4 times higher in the Burundian (270 pers. km^{-2}) and Rwandan (321 pers. km^{-2}) hills compared to the Ugandan (123 pers. km^{-2}) and Tanzanian (39 pers. km^{-2}) lowlands (density figures are for 2002) Historically, people preferred to settle in the Rwandan and Burundian part of the basin. Given the high dependency of the population on agriculture (90%, World Bank 2008; Kagera Basin Monograph, 2008), this most likely linked to the good climate and soil fertility, favourable for intensive agriculture. In this part of the basin, initially, problems of soil degradation and soil erosion were solved by long fallowing or migration and new land clearing. Following the population increase people began settling in marginal areas such as on steep hillslopes, poor soils, high altitude regions and pasture areas (Olson, 1990). At the same time, the land use intensified with reduced fallows, near-continuous cropping and labour-intensive management (Boserup, 1965, Clay and Lewis, 1990). These described changes are clearly reflected in the reported changes of this study, where initially most agricultural expansion is reported in the Rwandan and Burundian part of the basin, but subsequently shifts towards the eastern, more savannas dominated areas. Simultaneously with the increased demand for agricultural production, the demand for fuel wood can be expected to have increased, given that fuel wood accounts for 92% of the populations' energy demand (Kagera Basin Monograph, 2008), which is not surprising in an area where only 2.6% of the rural population has access to electricity (Kagera Basin Monograph, 2008; Musahara and Rao, 2009). Woodland degradation, possibly leading to conversion to a state that makes it appear as savannas from RS data is likely under these circumstances, providing a possible explanation for the observed savanna-woodland conversion.

Land use policies have been quite different between the four countries covering the basin. Whilst Burundi has hardly any official regulations and customary law dominates, the other countries have gone through a series of different land use policies, related to the colonial and political history of these countries. However, also some general observations can be made for the four countries. Colonization modified the traditional economic organisation by the introduction of cash crops produced for European markets. Major exports of the countries of the Kagera basin consisted (and still consist) of traditional exports, dominated by agricultural cash crop such as coffee, tea, cotton, pyrethrum and tobacco. Between 1960s and 1980s, farmers obtained cash from sale of coffee, cotton and tea. High production was achieved through expansion into virgin lands (Smith, 1984). Due to price fluctuations in the global market and decline in soil fertility, the production of traditional cash crops declined. At the same time the commercialization of other crops (e.g., maize, cassava, rice, banana, sweet potatoes) for local or regional created alternative sources of income for farmers. The production of these crops expanded into marginal lands and previously reserved land (Baijukya, 2004).

Besides these general political and economic drivers, as mentioned, some country specific historical changes can be addressed to explain the observed changes. For Burundi, this is the least straightforward, given its remaining dependence on customary law. Only recently (Burundi policy reform, 2008) the National Strategy of Environment in Burundi (SNEB) and the Code of the Environment are recommending harmonization of land legislation and forestry practices as well as the implementation of strategies to ensure sustainable use of natural resources. For Rwanda, during the colonial time, pastures (mostly savannas) were protected against agricultural expansion by cattle armies. However, after independence in 1961, this land was released for agriculture, facilitating the reported conversion of savannas into farmlands (Olson, 1994). After 1994 policies were put into place that enforced the planting of trees (mainly *Eucalyptus*) on communal lands and degraded farmlands, matching up with the observed increase in plantation forest. This increase in plantation forests was further enhanced by the increasing demand for firewood and construction wood, making wood an interesting commodity for farmers. Besides, the Rwandan government put conservation efforts into place to protect the remaining parts of the Nyungwe forest. This was partially achieved by planting a buffer of *Pinus patula* around the forest, also increasing the share of plantation forest. In Uganda, the land use policy inherited a stalemate from its colonial past until today. In the 1900 Uganda agreement, a situation was created where multiple rights and overlapping interests caused a situation where people own land they don't use, and other people use land they don't own. In an attempt to resolve this, the 1995 Constitution of Uganda, created a trust over specified natural resources (including pasture lands and forest reserves), to protect these for the common goods of the country. However, non-transparent policies, inadequate land use planning and weak policy enforcement have created a situation where local people have converted forests, woodlands and savannas into farmlands. While at the same time, economically focused policies led to large scale clearing of forests, woodlands and savannas for the production of sugarcane, cotton, coffee and tea (Kagera monograph 2008). The observed large scale conversion of savannas into farmlands between 1985 and 1995 in the Ugandan part of the basin seems understandable in the light of this political reality. In Tanzania, before independence (1961), land was a property of clans under the jurisdiction of chiefs (Rald and Rald, 1975). According to Milne, (1938) firm traditional laws restricted extensive cultivation of savannas, woodlands and primary forests. Out migration was a way to relieve population pressure (McMaster, 1960). After independence, all land in Tanzania was declared government property to enhance adoption of socialism (*Ujamaa*), and to give protection against socially disruptive tendencies such as land speculation and excessive aggregation from a rich minority (Nyerere, 1967). This policy was gradually abandoned in 1985 after the government had signed an agreement with the International Monetary Fund (IMF) to liberalize the economy. The 1995 national land policy gave a substantive push to economic and social development objectives under a free market economy. Combined with a poverty eradication strategy it has accelerated the conversion of common land for crop production (de SteenhuijsenPiters, 1999). This matches with the fact that a lot of the post 1985 conversions (Figure 2.3) fall within the Tanzanian part of the catchment.

2.4.2 Impacts

The effects of land cover conversion can be highly variable depending on the approach followed to convert from one cover type to another. Ecological effects of land cover conversion include changes in soil quality, soil erosion, water quality, biodiversity loss and habitat availability. As indicated by Tilman (1999) expansion of agriculture is expected to have most profound impacts on freshwater ecosystems. Aquatic nutrient eutrophication can lead to loss of biodiversity, outbreaks of nuisance species, shifts in the structure of food chains, and impairment of fisheries. These are all effects that have been previously reported for Lake Victoria (e.g. World Bank, 1996; Scheren et al. 2000, Verschuren 2002; Kagera monograph 2008; Musahara and Rao, 2009), and clearly, in the upstream part of the Kagera catchment these type of conversions have occurred as shown in this study. It can be questioned however to what extent these conversions led to eutrophication, and which macro nutrients were added most. The type of conversion and the economic setting under which the conversions occurred suggest that most conversions have been to farmland where low external input practices are applied. However, erosion and loss of sediment could also have detrimental effects on the water quality in the catchment. It remains to be seen what the quantities of these potential losses are, and whether they remain in the catchment or actually make it down to the lake. To date there are no conclusive studies for the nutrient and sediment inputs into lake Victoria from these upstream catchment areas, although this study clearly shows that land cover conversions were considerable in the past century.

Related to soil quality and nutrient status is the amount of soil organic matter (SOM) in the soils. Land cover change has been related to changes in SOM (Powers, et al. 2011). The magnitude of SOM stocks in the soil is central to soil quality and productivity because it maintains to a large extent many soil physical properties, provides substrate to micro biota and plays a role in nutrient buffering and supply (Nye and Greenland, 1960; Sanchez, 1976; Chapman, et al. 2003). Conversion to permanently cropped land, in the absence of intensive soil management, can lead to reductions in soil organic matter and nutrient depletion (Lal, 2009; Smaling, et al. 2012). Forested areas are especially related to high SOM stocks, and conversion of native forest to agriculture is expected to decrease SOM stocks considerably (Powers, et al. 2011). Not only will this have consequences for productivity, but also in the light of the global carbon cycle and global climatic change, quantifying the changes in SOM stocks would be relevant. Land cover changes are linked to approximately 20% of the global $CO2$ emissions to the atmosphere (IPCC, 2007; Van der Werf, et al. 2009). These studies suggest that most of these emissions are coming from the tropics (Rhoades, et al. 2000; IPCC, 2007; Verburg, et al. 2011). However, data on SOM changes in the tropics are scarce, especially for eastern central African landscapes, such as the Kagera Basin. Given the high level of land cover change that has happened, and is still happening in this region, it seems that studies addressing the effects of land cover change on the SOC and nutrient stocks in tropical African landscapes are profoundly needed.

Besides the effect of land cover changes on SOM stocks, also the changes in the above ground stocks can be considered substantial. As shown in this study, deforestation has been leveling off in the Kagera basin in recent years, but internal forest degradation, as well as conversion of woodlands into savannas has been continuing. The effect of this on carbon stocks in above ground biomass remains vague, although Nsabimana (2009) quantified part of these fluxes for the Nyungwe forest remnants in Rwanda; the role of plantation forests and also the role of savannas and woodlands in this equation requires further detailed studies.

Lastly, the repercussions of the quantified land cover changes for biodiversity remain largely unknown. According to the Convention on Biological Diversity (2010) the major pressures on biodiversity are habitat loss and degradation, climate change, excessive nutrient load and other forms of pollution, over-exploitation and unsustainable use and invasive alien species. As mentioned previously, the changes that occurred in the Kagera basin are most likely related to losses in nutrients that can cause nutrient loading in other locations. Likewise, the deforestation that took place in the beginning of the study period, and the ongoing forest degradation, relate to the habitat loss mentioned. However, detailed studies on the biodiversity status in the Kagera basin, and how this has been affected by land cover changes are so far lacking.

2.5 Conclusion

In this study, we have presented a spatially explicit LUCC analysis over the last 100 years of the Kagera river basin. The current state of land cover and its dynamics have most probably environmental implications in-terms of spatial environmental quality at the local and basin scale, however, to date most of the expected impacts remain unquantified for this basin. The results of this study offer therefore important directions for follow up research aimed at quantifying these impacts, including soil erosion, river sedimentation, soil organic carbon dynamics and soil quality changes, changes in above ground biomass, and lastly, biodiversity changes. Results of such studies are needed to feed into environmental monitoring plans and research models for future land cover scenarios and agricultural planning.

2.6 Acknowledgement

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Appendix 2.1 Supplementary materials

Table S2.1

Confusion matrix of land-cover classification map derived from the 1985 Landsat-5 TM image

Classified (land cover type)	Reference data											Total	Producer Accuracy	User Accuracy
	Dense Forest	Degraded Forest	Plantation Forest	Tea	Bamboo	Annual Crops	Savannas	Wood lands	Urban and Built-up	Water Bodies	Permanent Wetlands			
Dense Forest	12	2	0	0	0	0	0	0	0	0	0	14	71	86
Degraded Forest	5	6	0	0	0	0	0	0	0	0	0	11	75	55
Plantation Forest	0	0	24	12	0	0	0	0	0	0	0	36	80	67
Tea	0	0	6	48	0	0	0	0	0	0	0	54	80	89
Bamboo	0	0	0	0	55	0	0	0	0	0	0	55	100	100
Annual Crops	0	0	0	0	0	240	21	0	0	0	0	261	83	92
Savannas	0	0	0	0	0	48	140	40	0	0	0	228	87	61
Woodlands	0	0	0	0	0	0	0	104	0	0	0	104	72	100
Urban & Built-up	0	0	0	0	0	0	0	0	63	0	0	63	100	100
Water Bodies	0	0	0	0	0	0	0	0	0	100	0	100	91	100
Permanent Wetlands	0	0	0	0	0	0	0	0	0	10	88	98	100	90
Total	17	8	30	60	55	288	161	144	63	110	88	1024		
Overall Accuracy = 86														
Overall Kappa = 0.83														

Table S2.2
Confusion matrix of the land-cover classification map derived from the 2010 Landsat-7 ETM+ image

Classified (land cover type)	Reference data											Total	Producer Accuracy	User Accuracy
	Dense Forest	Degraded Forest	Plantation Forest	Tea	Bamboo	Annual Crops	Savannas	Permanent Wetlands	Urban and Built-up	Water Bodies	Wood lands			
Dense Forest	10	4	0	0	0	0	0	0	0	0	0	14	77	71
Degraded Forest	3	10	0	0	0	0	0	0	0	0	0	13	71	77
Plantation Forest	0	0	45	12	0	0	0	0	0	0	0	57	83	79
Tea	0	0	9	76	0	0	0	0	0	0	0	85	86	89
Bamboo	0	0	0	0	55	0	0	0	0	0	0	55	100	100
Annual Crops	0	0	0	0	0	162	21	0	0	0	0	183	93	89
Savannas	0	0	0	0	0	12	77	0	0	0	0	89	79	87
Permanent Wetlands	0	0	0	0	0	0	0	64	0	0	0	64	100	100
Urban & Built-up	0	0	0	0	0	0	0	0	63	0	0	63	88	100
Water Bodies	0	0	0	0	0	0	0	0	9	110	0	119	100	92
Woodlands	0	0	0	0	0	0	0	0	0	0	121	121	100	100
Total	13	14	54	88	55	174	98	64	72	110	121	863	100	100

Overall Accuracy = 92

Overall Kappa = 0.91

Table S2.3

Land cover conversion matrix of the Kagera Basin since 1901s' to 1974

land cover Change area between 1901 to 1974	1974										
1901	Farm lands (ha)	Bambo o (ha)	Degraded Forest (ha)	Dense Forest (ha)	Permanen t Wetlands (ha)	Plantation Forest (ha)	Savannas (ha)	Tea (ha)	Urban and Built-Up (ha)	Water Bodies (ha)	Wood lands (ha)
Bamboo (ha)	9714	**5015**	0	0	0	0	0	0	0	0	0
Dense Forest (ha)	114730	0	6764	**314458**	0	285	6048	692	0	0	0
Permanent Wetlands (ha)	0	0	0	0	**284266**	0	0	0	0	0	0
Savannas (ha)	112101 1	0	0	0	0	41595	**947379**	3125	1641	0	0
Water Bodies (ha)	0	0	0	0	0	0	0	0	0	**10740 1**	0
Woodlands (ha)	616463	0	0	0	0	25416	769467	1605 4	0	0	**1608521**

Table S2.4

Land cover matrix of Kagera Basin since 1974 to 1985

Land cover change area between 1974 to 1985	1985										
1974	Farmlands (ha)	Bamboo (ha)	Degraded Forest (ha)	Dense Forest (ha)	Permanent Wetlands (ha)	Plantation Forest (ha)	Savannas (ha)	Tea (ha)	Urban and Built-up (ha)	Water Bodies (ha)	Woodlands (ha)
Farmlands (ha)	1801534	0	0	0	0	71454	0	265	3929	0	0
Bamboo (ha)	0	5015	0	0	0	0	0	0	0	0	0
Degraded Forest (ha)	3089	0	3401	0	0	0	0	274	0	0	0
Dense Forest (ha)	87477	0	32557	157896	0	3535	12457	1042	0	0	19494
Permanent Wetlands (ha)	0	0	0	0	284266	0	0	0	0	0	0
Plantation Forest (ha)	23141	0	0	0	0	44053	0	102	0	0	0
Savannas (ha)	637388	0	0	0	0	33646	1051861	0	0	0	0
Tea (ha)	1649	0	0	0	0	0	0	2958	0	0	0
Urban and Built-Up (ha)	0	0	0	0	0	0	0	0	1641	0	0
Water Bodies	0	0	0	0	0	0	0	0	0	107401	0
Woodlands (ha)	350265	0	0	0	0	0	353952	0	0	0	904304

Table S2.5

Land cover matrix of Kagera Basin since 1985 to 1995

Land cover change area between 1985 to 1995	1995										
1985	Farm lands (ha)	Bamboo (ha)	Degraded Forest (ha)	Dense Forest (ha)	Permanent Wetlands (ha)	Plantation Forest (ha)	Savannas (ha)	Tea (ha)	Urban and Built-up (ha)	Water Bodies (ha)	Woodlands (ha)
Farmlands (ha)	2769767	0	0	0	0	123758	0	2816	8200	0	0
Bamboo (ha)	0	5015	0	0	0	0	0	0	0	0	0
Degraded Forest (ha)	262	0	17682	12463	0	3184	2052	315	0	0	0
Dense Forest (ha)	2082	0	32407	121845	0	62	1282	86	0	0	132
Permanent Wetlands (ha)	0	0	0	0	284266	0	0	0	0	0	0
Plantation Forest	99851	0	0	0	0	52575	0	82	180	0	0
Savannas (ha)	399903	0	0	0	0	18544	999824	0	0	0	0
Tea (ha)	0	0	0	0	0	0	0	4641	0	0	0
Urban and Built-up (ha)	0	0	0	0	0	0	0	0	5569	0	0
Water Bodies (ha)	0	0	0	0	0	0	0	0	0	107401	0
Woodlands (ha)	78504	0	0	0	0	2796	406825	0	0	0	435673

Table S2.6
Land cover matrix of Kagera Basin since 1995 to 2010

Land cover change area between 1995 to 2010	2010										
1995	Farmlands (ha)	Bamboo (ha)	Degraded Forest (ha)	Dense Forest (ha)	Permanent Wetlands (ha)	Plantation Forest (ha)	Savannas (ha)	Tea (ha)	Urban and Built-up (ha)	Water Bodies (ha)	Woodlands (ha)
Farmlands (ha)	3350237	0	0	0	0	132	0	0	0	0	0
Bamboo (ha)	267	4747	0	0	0	0	0	0	0	0	0
Degraded Forest (ha)	144	0	17348	30975	0	341	107	573	0	0	602
Dense Forest (ha)	3976	0	7136	123139	0	0	0	56	0	0	0
Permanent Wetlands (ha)	0	0	0	0	284266	0	0	0	0	0	0
Plantation Forest (ha)	215	0	0	0	0	199848	858	0	0	0	0
Savannas (ha)	256179	0	0	0	0	0	1153804	0	0	0	0
Tea (ha)	0	0	0	0	0	0	0	7939	0	0	0
Urban and Built-up (ha)	0	0	0	0	0	0	0	0	13950	0	0
Water Bodies (ha)	0	0	0	0	0	0	0	0	0	107401	0
Woodlands (ha)	4914	0	0	0	0	0	19517	0	0	0	411373

Chapter 3

Contemporary land use / land cover types determine soil organic carbon stocks in South-West Rwanda

Abstract

Soil organic carbon (SOC) constitutes a large pool within the global carbon cycle. Land use change significantly drives SOC stock variation. In tropical central and eastern Africa, how changes in land use and land cover impact on soil C stocks remains unclear. To address this knowledge gap, we classified the current and historical land cover and measured SOC stocks under different land cover, soil group and slope type in the humid zone of south-west Rwanda. It was observed that SOC levels were best explained by contemporary land cover types, and not by soil group, conversion history or slope position, although the latter factors explained partly the variation within annual crop land cover type. Under all soil groups, forest land uses held higher SOC stocks than agricultural land use types. Lack of the influence of land use history on SOC stocks suggests that after conversion to a new land use/ land cover, SOC stocks reached a new equilibrium within the timestep that was observed (25 years). For conversion to annual crops, SOC stocks reach a new equilibrium at about 2.5% SOC concentration which is below the soil fertility threshold of 3% in the region. SOC stock declined under transitions from banana-coffee to annual crop by 5% (6 Mg C ha^{-1}) or under transitions from natural forest to degraded forest by 21% (66 Mg C ha^{-1}) and increased for transitions from annual crops to plantation forest by 193 % (245 Mg C ha^{-1}). Forest clearing for agricultural use resulted in a loss of 272 (72%) Mg C ha^{-1}. Assuming steady states, the data can also be used to make inferences about SOC changes as a result of land cover changes. We recommend that SOC stocks should be reported by land cover type rather than by soil groups which masks local land cover and landscape differences. This study addresses a critical issue on sustainable management of SOC in the tropics and global carbon cycle given that it is performed in a part of the world that has high land cover dynamics while at the same time lacks data on land cover changes and soil organic carbon dynamics.

Keywords: *land use change, land degradation, soil organic carbon, soil fertility, soil C sequestration, global climate change*

3.1 Introduction

The atmospheric concentration of CO_2 and other radiatively active gases is steadily increasing in the last century due to anthropogenic activities (Lal 2001; IPCC 2007a). These increased concentrations are thought by the IPCC to pose a threat by causing global warming and a cascade of derived alterations in the earth climate system (IPCC 2007a). To alleviate the threat posed by increasing CO_2 emission to the atmosphere, the global community has focused their attention to the management of the carbon (C) cycle (Batjes 1996; Su et al. 2006). A number of studies link land use and land cover changes (LUCC) to approximately 20% of the global CO_2 emissions to the atmosphere (e.g., Houghton 1995, 1999, 2005; Batjes 2004; IPCC 2007b; Van der Werf et al. 2009). Other studies suggest that most of these emissions are coming from the tropics (e.g., Rhoades et al. 2000; IPCC 2007b; Verburg et al. 2011). These emissions are derived from two different sources, above ground biomass and soil organic matter (SOM), of which the latter has a lag in response to LUCC related to the decomposition rate of SOM. Soil organic matter contains approximately 58% carbon, a value that can be used to convert SOM to Soil Organic Carbon (SOC). Losses in SOM are not only having repercussions on the global climatic system but also heavily influence more directly local soil processes. The magnitude of SOM stocks (and thus SOC stocks) in the soil is central to soil quality and productivity because it maintains to a large extent many soil physical properties (e.g. aeration, water retention, formation of stable aggregates) (Lal 2004), provides substrate to micro biota (Nye and Greenland 1960; Sanchez 1976) and plays a role in nutrient buffering and supply (Lal 2004; Smaling et al. 1993; Young 1997; Schnitzer 1991; Chapman et al. 2003; Smaling et al. 2012). Levels of SOM have been shown to be particularly important in cultivated tropical soils where high temperatures lead to rapid SOM breakdown, while organic matter reserves are already low and the use of other inputs for soil fertility management is rare (McDonagh et al. 2001). Most agricultural soils in the tropics have lost 20% to 83% of their original SOC over a period of 5 to 50 years of cultivation (Detwiler 1986; Schlesinger 1986; Vagen et al. 2005, Esteban et al. 2000; Lal 2007, 2009), especially in sub Saharan Africa, affecting plant nutrient supply or soil fertility and consequently food production and security (Detwiler 1986; Schlesinger 1986; Murty et al. 2002; Vagen et al. 2005, Lal 2004, 2009; Smaling et al. 2012).

Policy makers across the tropics propose that carbon finance could provide incentives for forest frontier communities to move away from agricultural land use to other systems that potentially reduce emissions and/or increase carbon sequestration. However, it is unclear what exactly controls tropical SOC stocks. There is little certainty regarding the carbon outcomes of many key land-use transitions at the center of current policy debates. Besides, it is difficult to obtain reliable estimates of the historic loss of the SOC pool in the tropics (Lal 2002). Data on changes in SOC stocks in tropical land use systems is scarce or fragmented (e.g., Guo and Gifford 2002; Minasny et al. 2010; Powers et al. 2011; Eclesia et al. 2012) and consensus has yet to be reached on how much carbon is lost by changes in tropical land use systems (e.g., Fearnside and Laurance 2003; Ramankutty et al. 2007, Van der Werf et al. 2009; Phelps et al. 2010; Ziegler et al. 2012; Eclesia et al. 2012). A meta-analysis of 74 publications on the influence of land use changes on SOC stocks by Guo and Gifford (2002) showed that most data that could be found were drawn from only four countries (Australia, Brazil, New Zealand, and USA) indicating possible bias in findings on the effects of LUCC on SOC stocks. A more recent but similar study by Powers et al. (2011) covering 27 countries from 80 different studies showed a more even spread over the world, but still with a very small coverage of Africa. No reports were mentioned on the tropical landscapes of central Africa.

Given the high level of land cover change that has happened, and is still happening in this region (Wasige et al. 2013), it seems that studies addressing the effects of land cover change on the SOC in tropical African landscapes are profoundly needed. There has been some attempts to quantify SOC stocks in this region: Batjes, (2008) presented SOC estimates per agro-climate zone and soil type for Central Africa (CAF) based on the soil and terrain (SOTER) database at a 1:2.000.000 scale. Gaps in the data were filled using expert rules and taxonomy-based pedotransfer procedures for similar groups of FAO (1988) soil types, providing a first coarse scale approximation of SOC stocks for CAF. This approach has limitations on a smaller scale, particularly in regions with large variations in topography, climate and land use types. Variability in the existing data is typically explained by soil and climate factors with little consideration given to land use and management history. Verdoodt et al. (2010) reported baseline SOC stocks of Rwandan topsoils (0-30 cm) calculated for 121 profiles, representing 99 soil series which spatially cover 47% of the Rwandan soilscape on soil map sheets at a scale of 1:50,000. Average sampling density was 1 profile for every 14 km^2. About 6% of the profiles were located within a natural reserve. The results roughly correspond to the Central African estimates of Batjes (2008) and faced similar limitations such as coarse resolution of sampling, aggregated forest cover to one unit of forest instead of natural forest, degraded nature forest and plantation forest. Aggregation was also performed for crop cover into one unit instead of annual crop, banana-coffee and tea crop. More recently, a study on carbon stocks in the Nyungwe National Forest (Rwanda, part of CAF) focused on a relatively undisturbed area of the forest (Nsabimana 2009). It was unclear from this study how internal forest degradation or land cover conversion influenced SOC stocks. Yet, this is an area that experienced considerable LUCC in the recent past, including conversions from primary forest into smallholder farming systems with very limited external inputs (Wasige et al. 2013). This raises questions whether changes in SOC stocks in this region are occurring under the current LUCC, and what the magnitude of these changes are. Therefore, the main objective of this paper is to study the relation between land use, its change, and soil organic carbon stocks. In order to achieve this, land use has been classified using satellite imagery of 1986, 1995 and 2010. The changes between these two time periods (1986-2010 and 1995-2010) were established as well. The land units identified were overlaid with soil classification units taken from secondary information (Verdoodt and Van Ranst 2003). Soil organic carbon was measured, and after correction for clay mineralogy, correlated with current land use, land use change between 1986 and 2010, and between 1995 and 2010, slope positions in the landscape and soil classification units. The main question addressed was; which of the mentioned factors has an effect on SOC stocks?

3.2 Materials and methods

3.2.1 Location and description of study area

The study site is the 506 km^2 river Rukarara catchment (Figure 3.1) in the humid hill slopes of South-West Rwanda and in the upper reaches of the Nile Basin ($2^0$28′-$2^0$34′ S and 29^0 23′-$29^0$29′E). The part of the catchment that falls within Nyungwe forest is a globally important biodiversity spot, and a water tower of the Nile basin (World Bank 1996). The hill slopes exhibit physical variability in terms of elevation, climate, geology, and soils as the river Rukarara drains and descents the catchment. The river has its source at about 3000 metres above sea level (m.a.s.l) and descends to 1470 m at the outlet. The steep slopes have lost most of their forest cover and are intensively cultivated by smallholder farmers. Plantation forests are replacing annual crops on highly degraded soils and steep slopes where cultivation is no longer possible. The low land is hotter and drier than the upstream area with temperature ranging from 15^0C - 25^0C. The rain pattern is bi-modal, with long rains occurring during September to January, and shorter rains in March to June. Annual precipitation ranges from 1200 to over 1800 mm per annum. The catchment experiences high rainfall intensities. Rain falls mainly during storms producing large amounts of runoff that consequently often acts as a driver of soil erosion (Verdoodt and Van Ranst 2003; Bizoza and De Graaff 2010). The geology of the area consists of bedrock formed in the middle Precambrian period and is differentiated in two parts of the catchment. In the eastern part of the catchment, the substratum is derived from granites and granitic rock. In the western part of the catchment, schist, quartzite and dolerite are the main parent rocks. A soil map (1:50000; FAO 1988 classification) of the area is available (Verdoodt and Van Ranst 2006) and reveals that the soils are variable; Acrisols, Alisols, Ferralsols, Gleysols, Regosols, Histosols and Cambisols occur. The vegetation consists of a fully forested upper part of Nyungwe natural forest while the middle to lower parts of the catchment have agricultural crops. The main cash crops are tea *(Camellia spp.)* and coffee (*Coffea canephora*), and food crops are maize (*Zea mays*), banana (*Musa spp.),* beans (*Phaseolus vulgaris*), cassava (*Manihot esculenta*), sweet potatoes (*Ipomoea batatas*), solanum potatoes (*Solanum tuberocum spp*), sorghum (*Sorghum Bicolor (L.) Moench*) and wheat (*Triticum*). Food crops are typically grown on plots of less than 1 ha. Cultivation and tillage by smallerholder farmers is mainly undertaken with the use of simple handheld implements. Some farmers construct terraces and contour ridges to minimize the effect of soil erosion along the slopes but they are poorly maintained, still resulting in erosion problems. Both crop rotation and mixed farming are practiced by the majority of the farmers. In most cases, organic manure from livestock pens mixed with crop residues are applied in seedling holes when planting seeds to increase soil fertility which means that SOC will not be uniformly distributed in the agricultural fields. There is also a high degree of crop diversification and intercropping. Due to the high economic value of tea, some farmers have replaced their traditional food crops with tea. Tea production started around 1969 and today most tea farmers are organized into community tea groups.

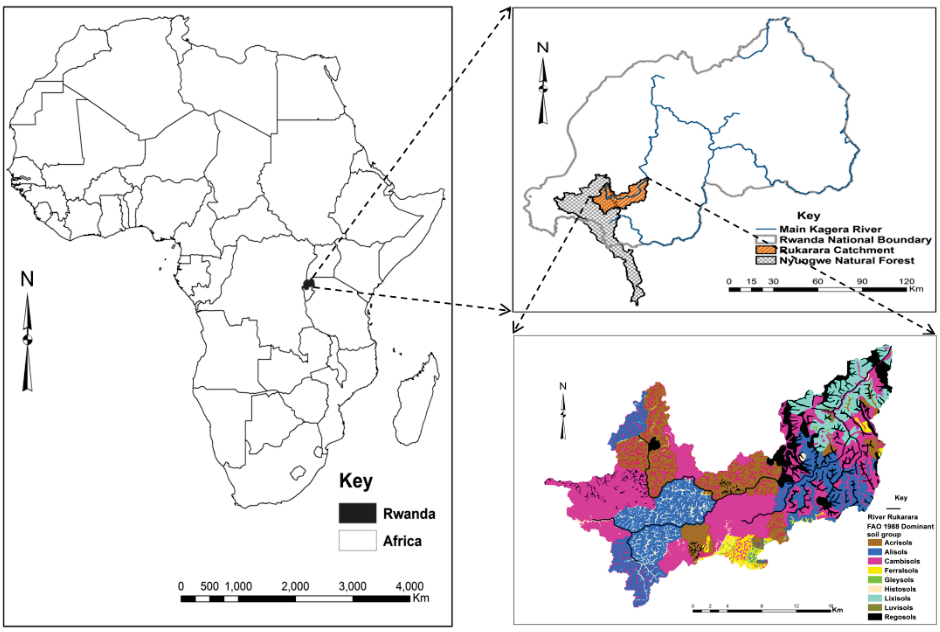

Figure 3.1: Location of Rwanda and FAO soil group map of Rukarara catchment

3.3 Research Steps

3.3.1 Land Cover Classification and change detection

Landsat images of path 173 and row 62 that are predominantly cloud-free for the moments; 1986-July-19th, 1995-January-17th and 2010-August-22nd were selected for land cover classification and change detection. The USGS and IGBP land cover classification system (Anderson et al. 1976; Belward 1996) were adopted and modified to include classes that are applicable to the study area and general enough so that they can be readily identified on Landsat imagery. Existing land cover types in this area include; natural forest, degraded forest, plantation forest (which are mostly with fast growing species such as eucalyptus), savannas, annual crops, banana-coffee plantations and tea plantations. Table 3.1 provides a descriptive summary of the used land cover classes for this study. Landsat images were geo-referenced and rectified. Geo-coded images were then overlaid with vector data (roads, rivers and lakes) where a perfect match was observed and then sub-setted to the extents of the study area. Supervised classification procedures were implemented using Erdas 2010 image processing software to classify the Landsat images into established land cover classes using the Maximum Likelihood Classification (MLC) algorithm. The final classification and interpretation of the 2010 land cover map was based on field observations from 2010 that were used as reference information on land use classes and for the accuracy assessment. Further details of the methodology and the accuracy of the results are described in Wasige et al. (2013).

Table 3.1: Description of land use/land covers classes and definitions

Land cover classes	Definitions
Natural Forest	Native forest before it's cleared for other land uses. Mature vegetation >5m; >60% ground surface covered by trees; almost all trees remain green all year; canopy is never without green foliage
Degraded Forest	Part of the natural forest degraded by selective logging and fire. Includes coniferous, deciduous and mixed forests. Mature vegetation >5m; > 40 % to <60% ground surface covered by trees
Plantation Forest	Manmade forest developed by planting faster growing trees (Pinus patula and Eucalyptus)for individual land cover types ; >40% ground surface covered by trees
Savannas	Shrubs <2m; shrub canopy coverage <40%; remaining cover either bare or annual herbaceous/ grasses; shrubs either evergreen or deciduous
Tea	The tea-plant, in the natural state, grows into a small or medium-sized tree, but it is pruned and trained to knee height (< 0.5m) form a many-branched low bush mart
Banana-Coffee	The banana plant is the largest herbaceous flowering plant. The plants are normally tall and fairly sturdy that grows 6 to 7.6 metres tall, growing from a corm. Leaves are spirally arranged and may grow 2.7 metres long and 60 cm wide. Plant canopy cover is >10% and <80%. The coffee plant is a woody perennial evergreen dicotyledon that belongs to the Rubiaceae family. The plant can grow to heights of 10 meters if not pruned; tree canopy cover >10% and <60%.
Annual Crops	Land cultivated for food and fiber; >80% of landscape covered in crop-producing fields and includes cultivated land without crops

3.3.2 Soil Sampling and laboratory analysis

Stratified toposequential randomized sampling scheme was used to select soil sampling sites based on a sampling matrix of contemporary (i.e. 2010) land cover types and three slope positions. Slope positions were categorized as; foot slope, mid slope and upper slope. The upper slope position includes upper and lower interfluves and receives little or no runoff, but contributes runoff to lower slope positions. The middle slope position includes shoulder, upper and lower linear slopes, and receives runoff from the upper slope and contributes runoff to foot slope positions. The foot slope represents the base of the hill (Wang et al. 2001; Kagabo et al. 2013). Slope positions were derived from an ASTER DEM (30 m x 30 m) using the Topographic Position Index (TPI) tool in ArcGIS10. The output raster was overlaid with the land cover map of 2010 (Figure 3.2) to identify sample strata in ArcGIS10. Sampling points were created randomly using ArcGIS10 software. A field sampling map with GPS points was then printed to support field identification of selected sampling points. A handheld GPS was used to locate these positions in the field. A total of 644 sites were visited for the collection of soil samples during the month of October 2010, with a soil auger at two depths (0- 20 cm and 20 - 50 cm) in two different fieldwork campaigns. During the field campaign, 87 soil samples were collected from Nyungwe forest and 557 samples were collected from the agricultural part of the catchment. For each site, a plot of 10 m x 10 m was outlined using a tape measure. Four spots along each of the diagonals and 1 point from the centre per site were sampled. Soil samples from each plot were thoroughly mixed to make a composite sample which was then quota sampled for about 500 g of soil and taken to the laboratory for chemical analysis.

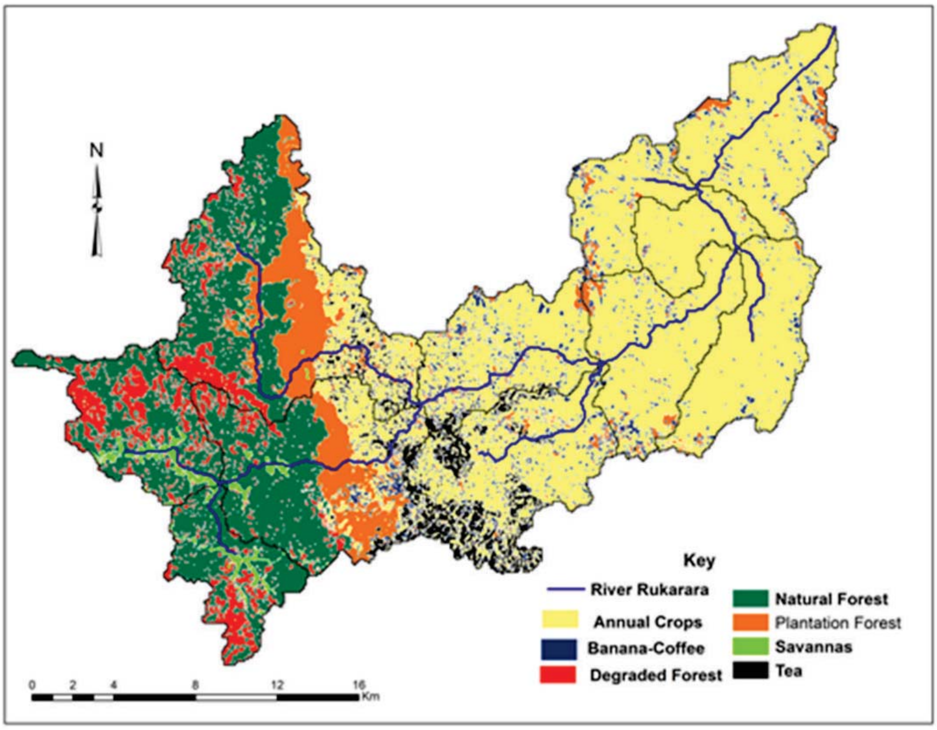

Figure 3.2: Land cover types of Rukarara catchment, south west Rwanda during 2010

In the laboratory, soil samples were analysed by the Walkley Black method was used to determine soil organic matter (SOM) content. The amount of carbon in SOM varies, but often the "Van Bemmelen Factor" of 1.724 is applied (Schumacher 2002), based on the assumption that 58% of SOM consist of soil organic carbon (SOC). Although higher values have been suggested (up to 2.5; Broadbent 1953) in this study we used the more commonly used value of 1.724. The Walkley Black method is commonly used for routine SOC assessments (Bell and Lawrence 2009; Tornquist et al. 2009), and involves a wet combustion of the SOM with a mixture of potassium dichromate and concentrated sulphuric acid at about 125^{0}C. The residue dichromate is titrated against ferrous sulphate and a correction factor of 1.3 is applied in the calculation of the results to compensate for the incomplete destruction of the organic matter during the process (Van Reeuwijk 2006).

About 2 soil bulk density samples at 0 – 20 cm and 20 - 50 cm soil depth were randomly collected in mini pits using PF-rings located at the middle of the various plot sites. After drying the samples in an oven at 105^{0}C, bulk density was calculated using standard procedures described in Anderson and

Ingram (1993). Calculation of SOC stocks followed the IPCC procedure (IPCC 2003) as presented in equation 1;

$$SOC_{site} = \sum_{layer=1}^{layer=j}(SOC_{content} * bd * (1 - cf) * sd * 10) \qquad \text{Eq. 3.1}$$

where; SOC_{site} is the soil organic carbon stock (Mg C ha^{-1}); $SOC_{content}$ is the concentration of soil organic carbon C (g C kg^{-1} of soil) for layer j of sampled soil depth; *bd* is the bulk density of an undisturbed volume of the soil (g cm^{3}); *cf* is the coarse fraction, i.e. the >2 mm coarse fragment (weight %) in a single sampled soil depth; and *sd* is soil depth, the thickness of the soil layer sampled (m).

3.3.3 Quantifying effect of LUCC

To measure changes in SOC as a result of LUCC, repeated sampling of plots before and after a land-use change occurs (chronosequence) would be desirable (Schlesinger 1986; Powers 2004). However, data from long-term monitoring of SOC stocks is often missing in the tropics, as is the case in the Rukarara catchment. This can be circumvented by a paired-comparison approach (biosequence), where sampling occurs at the same time on sites with similar soil and climate types but different land uses for comparison. It involves substituting space-for-time in field sampling of SOC stocks, comparing areas under native and changed land use and land cover. This approach holds the implicit assumptions that (1) the sites were similar in soils and climate apart from land use and (2) were in equilibrium prior to conversion (Rhoades et al. 2000; Powers 2004; Yemefack et al. 2006; Smaling et al. 2012). The latter assumption was implicitly made in this study, but the former was controlled by comparing sites with different land use histories as derived from the LUCC inventory. The percentage of SOC change was estimated according to equation 2 (Eclesia et al. 2012);

$$SOC\ change\ (\%) = \left(\frac{SOC\ current - SOC\ original}{SOC\ original}\right)\ x\ 100 \qquad \text{Eq.3.2}$$

where; SOC change is the percentage change in SOC after vegetation replacement, SOC current is the SOC content (Mg ha^{-1}) in the soil under the new vegetation, and SOC original is the SOC content (Mg ha^{-1}) of the native ecosystem.

3.3.4 Data processing and Statistical analyses

R software (R Development Core Team 2012) was used during data processing and analysis to detect changes in SOC stocks and concentrations. An Analysis of Variance (ANOVA) with a Games-Howell post hoc test was used to detect significant differences among land cover types, land cover conversion types, slope positions and soil groups. The Games Howell test

(Games and Howell 1976) was selected for the Post Hoc comparison because (1) it is robust for small unequal sample sizes and not all combinations of soil type and land cover type are there, and (2) it is robust for differences in variances across the groups, which was the case with our data set. We tested differences in SOC concentrations (in %) over the various explaining factors for both the top soil (0 – 20 cm) and sub soil (20 – 50 cm) layer separately, and for the SOC density (Mg C ha^{-1}) for the two layers combined (0 – 50 cm).

3.4 Results

3.4.1 Soil organic carbon by contemporary land cover types, soil types and slope position

Total carbon stocks for the top 50 cm layer is depicted in Figure 3.3a and b, showing that the highest SOC stocks can be found in forest cover types, while low SOC stocks can be found in agricultural cover types. The exception are plantation forests, which are in essence man-made, and often are covered by exotic (but fast growing) species, which have the highest SOC stocks. For all soil groups, SOC stocks under all forest land cover types combined was significantly different from that of agricultural land cover types (tea, banana-coffee and annual crops; $P<0.05$). The SOC stock variability for similar soil types, suggests that land cover causes the difference (figure 3.3b). Among forest cover types, SOC stocks were significantly higher for plantations compared to degraded forest cover types. There were no significant differences between natural and degraded forest SOC stocks. The highest SOC stocks in soils under forest cover were observed in Acrisols and Histosols which were significantly different from lower stocks in Cambisols and Alisols (Figure 3.3b). In the agricultural land cover type, high SOC stocks were observed in Ferralsols, under tea and in Lixisols under banana-coffee and low SOC stocks were observed in annual crops on the Acrisols (Figure 3.3b). Tea indeed has the highest SOC content for any of the agriculture-soil type combinations. On Cambisols, Alisols and Acrisols both forest and agricultural land cover types were observed, and for these soils also the largest spread in SOC values was found. Within Cambisols, SOC stock variability followed a similar trend where SOC in agricultural cover types was lower than in forest cover types. Figure 3.4 shows the variation of SOC stocks along slope positions for the different land cover types. Although differences were found between land cover types, marginal significant differences ($P<0.04$) in SOC stocks were only observed between the upper and lower slope positions in annual crops.

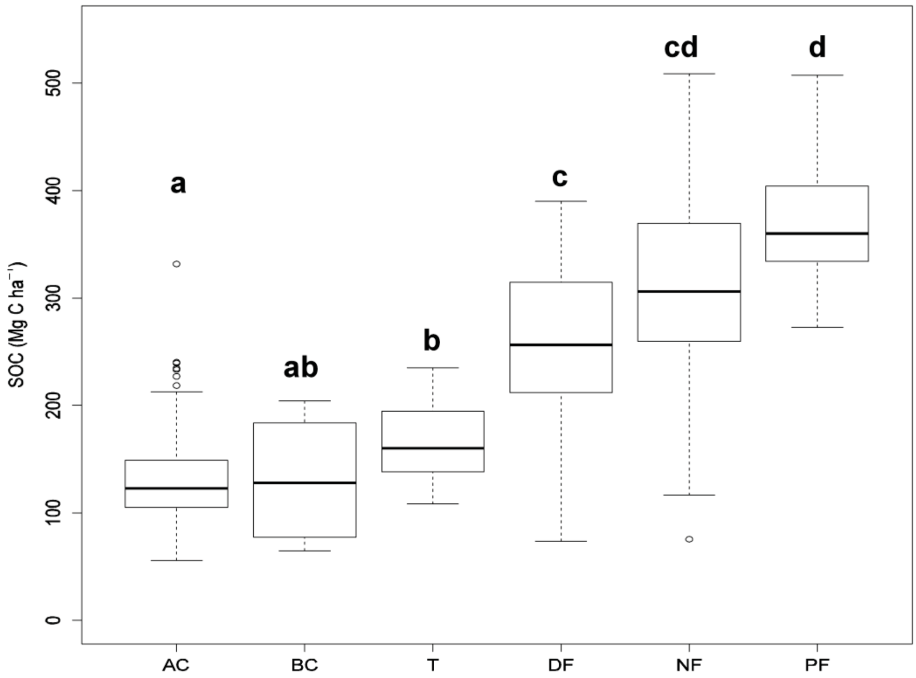

Figure 3.3a: SOC stocks for land cover type for 0-50 soil depth. Abbreviations: AC – Annual Crops; BC – Banana Coffee; PF – Plantation Forest; T – Tea; DF – Degraded Forest; NF – Natural Forest. Different letters indicate significant differences (P<0.05) according to the Games-Howell

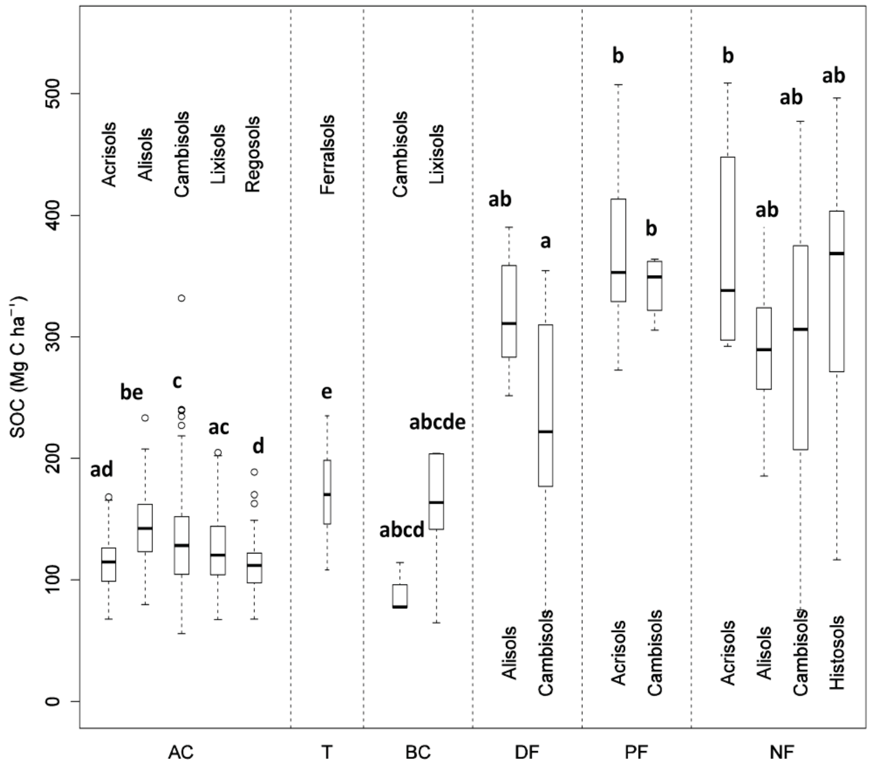

Figure 3.3b: SOC stocks for land cover type and soil group for 0-50 soil depth. Different letters indicate significant differences (P<0.05) according to the Games-Howell test for agriculture (AC, BC and T) and forest (DF, NF and PF) land cover types

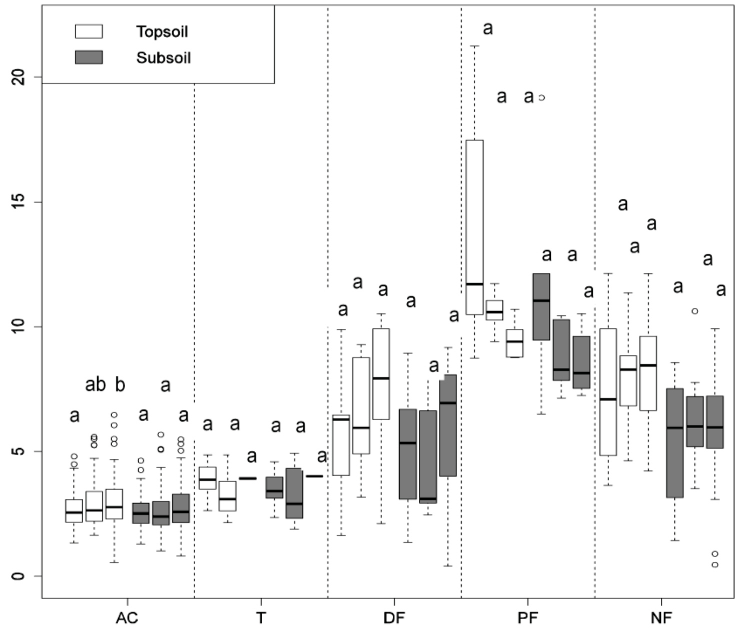

Figure 3.4: Variation of SOC stocks along three slope positions per land cover types. The first box of each series is down slope, the second mid slope and the last upslope. Different letters indicate significant differences (P<0.05) between slope positions within each land cover type according to the Games-Howell test

3.5 Soil organic carbon by land cover conversions

The different SOC concentrations associated with land cover conversion types between 1986, 1995 and 2010 are presented in Figure 3.5a and b. Similar to the above results, SOC concentrations are generally higher for the land cover types related to forestry, compared to land cover types related to agriculture. Between land cover transitions, areas that are currently under plantation forest, but originated from agriculture in 1986 have a significantly higher level of SOC concentrations compared to land cover conversion types from banana-coffee to annual crop. There were significant differences between land cover types that remained annual crop and that changed to annual crop from banana-Coffee, or between natural forest cover and land cover type that was originally natural forest and changed to degraded forest. No significant difference was observed between land cover changes that changed from banana-coffee to annual crop and annual crop to banana-coffee. Between 1995 and 2010 (see Figure 3.5b), there was no significant difference between land cover changes from banana-coffee to annual crop. Land cover change from plantation forest to annual crop had significantly higher SOC concentrations in the top soil compared to land cover that remained annual crop. SOC stocks increased for land cover transitions from annual crop to plantation forest by 193 % (245 Mg C ha^{-1}) while forest clearing for agricultural use resulted in a loss of 272 (72 %) Mg C ha^{-1}. Figure 3.5a shows that SOC concentrations declined for the land cover transition from banana-coffee to annual crop by 4.5 % (6 Mg C ha^{-1}).

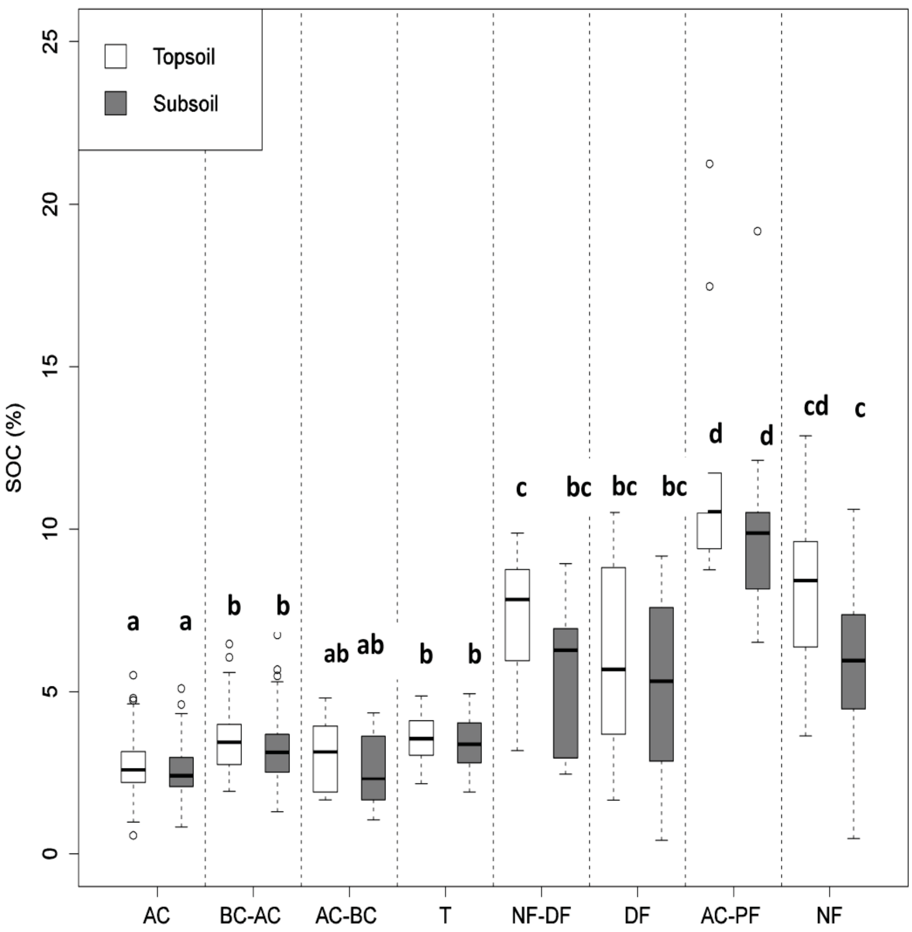

Figure 3.5a: SOC content (%) for the top soil (0-20 cm) and sub soil (20 – 50 cm) for land cover conversion types between 1986 and 2010. The cases where no land cover change occurred are indicated with one land cover type, while for the other cases, the first land cover type refers to the situation in 1986 and the second refers to 2010. Abbreviations: AC – Annual Crops; BC – Banana Coffee; PF – Plantation Forest; T – Tea; DF – Degraded Forest; NF – Natural Forest. Different letters indicate significant differences ($P<0.05$) according to the Games-Howell test for individual land cover types. Conversion types were compared amongst each other within one soil layer

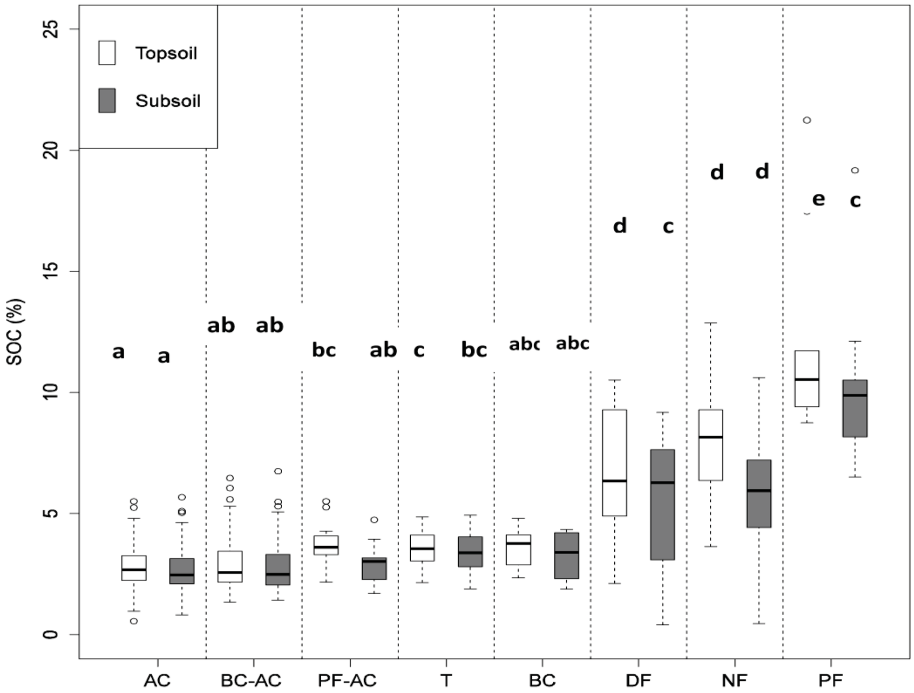

Figure 3.5b: SOC content (%) for the top soil (0-20 cm) and sub soil (20 – 50 cm) for land cover conversion types between 1995 and 2010. The legend is similar to Figure 3.5a

3.6 Discussion

3.6.1 Impact of contemporary land cover type on spatial distribution of SOC stocks

Generally, under all soil types, higher SOC stocks were associated with forest land uses and lower SOC stocks were associated with agricultural land uses. Natural forest and plantation forest had an average SOC stock of 310 Mg C ha^{-1} and 372 Mg C ha^{-1}, respectively, across soil types which is higher compared to similar studies for the same soil depth (0 – 50 cm) in Cameroon (85 Mg C ha^{-1}; Kanmegne 2004) and the Ethiopian highlands (190 Mg C ha^{-1}; Lemma et al. 2006). The actual value for each forest in different regions most likely varies with soil type, local land cover history and degree of deforestation.

Average SOC content in plantation forest was found to be higher compared to SOC stocks in natural forest. As pointed out by Lugo and Brown (1993) the differences in SOC accumulation under different tropical forest is due to differences in; tree species, litter production, litter quality, and microclimate.

Similar to finding by Mann (1986), our results suggests that afforestation with fast-growing (exotic) tree species leads to increases in SOC storage.

The SOC concentrations observed in annual crops was at the lower end of the spectrum at about 2.5 % SOC which is below the soil fertility threshold of 3 % for region (Okalebo et al. 2002) implying that more additions of soil organic matter and NPK fertilizers will be required to help increase biomass and thereby possible returns of organic matter to the soil. The SOC stocks observed in annual crops (127 Mg C ha^{-1}) is higher but in the same order of magnitude compared to values for the humid forest zone of southern Cameroon, where for the top 50 cm under different land uses along a slash-and-burn chronosequence, values between 61 and 100 Mg C ha^{-1} were reported (Kanmegne et al. 2004). The differences between the two study sites could be associated with differences in soil type and land use management. We do not have bulk density values from Kanmegne et al. (2004) study (a parameter which is included in the calculation when computing SOC stocks see Equation 1) but the slightly higher values of SOC in this study could be due to high bulk densities. The soils in the south west of Rwanda have been continuously cultivated for over 30 years (Wasige et al. 2013) and have poor soil structure (Verdoodt and Van Ranst 2003). Soils with poor soil structure will have high bulk density compared to soils with aggregate soil structure. Continuous cultivation without compensating for lost SOC and nutrients has been shown to be detrimental to soil structure (Lal 2004).

3.6.2 Interactive impacts of land cover type, soil type and slope position on SOC stocks

In our analysis, SOC levels were best explained by contemporary land cover types, and only in second instance by soil group, conversion history or slope position (Figures 3.3a and b). This is suggesting that under the same climate, SOC stocks are mainly land use specific rather than soil type specific (Figure 3.3b). We found that historical land cover had a marginal influence on the current SOC stocks (Figure 3.5a). This has been described before for plantation and pasture establishment in native forests and grasslands of South America (Eclesia et al. 2012) where SOC changes (0–20 cm of soil depth) were independent of the initial native vegetation (forest, grassland, or savanna) but strongly depended on the characteristics of the new vegetation (tree plantations or pastures), its age, and precipitation. Detwiler (1986) in the tropics and Esteban et al. (2000) in a global study, showed that the carbon stock reduced to approximately 60 % of the level under forest cover after 5 year of cultivation and to approximately 25% after 20 to 50 years. Generally, it is reported that conversion of forest to farmland in tropical regions may result in a 20 % to 83 % loss of SOC stocks (Detwiler 1986; Schlesinger 1986; Vagen et al. 2005; Lal 2009). Likewise, over a period of 25 years, soils in our study seem to have lost 72 % after conversion from forest to annual crops. Histosols under forest cover had the highest SOC stocks, which concurs with the fact that by definition these soils should have far more SOC being derived from incompletely decomposed plant remains. Under agricultural land cover types, the highest SOC stocks were observed in Ferralsols, but this observation is compromised with the fact that we only had tea plantations on this soil type. The high SOC stocks in Ferralsols under tea crop could be explained by a higher mineral fertilizer application enhance high biomass production and the practice of pruning tea and leaving the residues on the top soils as mulch, a common practice for this land cover type in Rwanda, which adds SOM to the soil (MINAGRI 2008).

Although Poels (1989) and Stoorvogel et al. (1997) have reported that even under natural conditions soil losses occur in natural forest, the carbon cycle in natural forest may be almost closed and the soil-plant system is often in a steady-state condition: SOC loses through decomposition are balanced by inputs from forest litter. This probably explains in part the luxuriance of SOC stocks in soils under forest cover (Yemefack 2005). Low SOC stocks observed under agricultural soils could be explained by poor soil fertility management and a long history of cultivation that exposes SOC to high temperatures and therefore high decomposition rates. In most cases farmers in Rwanda do not compensate for the carbon losses (Clay et al. 1998; Ansoms and McKay 2010). They may not have, or have limited access to manure and fertilizers and burn crop residues prior to seed bed preparation (Ansoms and McKay 2010). Besides, some studies have shown that the decomposition of tropical crop residues is normally faster than forest litter which would suggest that the potential they have to contribute to long-term SOC is limited (Jenkinson and Ayanaba 1977). Crop organic resources contain more of the labile SOC pools (light fraction C and particulate organic carbon) which are sensitive to management practices (Janzen et al. 1992; Solomon et al. 2000) and consequently highly influenced by the cultivation history of the soil (Nye and Greenland 1960; Davidson and Ackerman 1993; Murty et al. 2002). Studies have found that the light fraction and particulate organic carbon, are relatively easily decomposable and are greatly depleted upon cultivation (Cambardella and Elliott 1992; Six et al. 1999; Solomon et al. 2000) explaining why a new, lower equilibrium SOC stock is reached upon conversion from forest to agriculture. Very large additions of organic matter are generally required to compensate for the high decomposition rates and with that to increase the soil's carbon content (Hartemink 1997).

We also present a comparison between our results with course resolution data on SOC stocks at 0-50 cm soil depth as reported by Batjes (2008) for his study in Central Africa (CAF), which includes Rwanda (table 3.3). Generally, mean SOC stocks of soils for the humid zone of central Africa are comparable with those found in soils in this study, with marginal differences, except for Histosols, for which we only had data from forested land cover types. SOC values for soils under forest land cover were higher and under agricultural land cover were lower than the figure reported by Batjes (2008). The differences could accrue from the fact that for our study we corrected for land use/ cover impacts on SOC stocks which was not the case with Batjes study. We therefore point out that there is a need to correct for land use effects on SOC stocks at a scale relevant for local management as well as national carbon inventories and when results are to be compared with studies elsewhere.

Table 3.3: Current mean SOC by soil type and contemporary land uses

Soil types	Current mean SOC by soil type and contemporary land uses in Rukarara river catchment by 2010 at 0-50 cm soil depth			Difference between this study and the 176 Mg C ha^{-1} mean in 0-50 cm soil depth of Batjes (2008) for humid zone of Central Africa dataset that includes Rwanda		
	All mean of land cover types (Mg (C) ha^{-1})	Forest land uses (Mg Cha^{-1})	Agriculture (Mg Cha^{-1})	Forest land uses (Mg C ha^{-1})	Difference in mean soil SOC (Mg Cha-1)	Difference in mean soil SOC of Agriculture land cover (Mg Cha-1)
Acrisols	188	375	114	199	12	-62
Alisols	172	334	129	158	-4	-47
Cambisols	156	295	121	119	-20	-55
Histosols	307	487	157	311	131	-19
Ferralsols	169		169		-7	-7

Slope position was only important in annual crops land cover types and partly explained the variation in SOC stocks between upper and lower slope positions. Our finding is similar to Hancock et al. (2007) who reported that there was little relationship between hill slope position and SOC. Lal (1986) explained that soil erosion on hill slopes of agricultural catchments is one of the driving factors of SOC variability.

3.7 Conclusion

We evaluated the effects of land cover changes on SOC stocks in south-west Rwanda in tropical central Africa. It was observed that SOC levels were best explained by contemporary land cover types, and not by soil group, conversion history or slope position, although the latter factors explained partly the variation within land cover types. Generally, SOC stocks in the current agricultural land cover are at lower end of the spectrum at about 2.5 % SOC (or 127 Mg C ha^{-1}) which is below the soil fertility threshold of 3%, implying that additions of SOM are required to increase the soil's fertility. The low levels in SOC stocks in agricultural land use suggest a high potential for

SOC sequestration, when more attention would be paid to this in agricultural management. The higher SOC levels under tea cover suggest indeed that different management can partly induce this. We recommend that SOC stocks should be reported by land cover type rather than by soil groups because our results are better explained by the latter classification.

3.8 Acknowledgements

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Chapter 4

A new spatially explicit hydrological model for erosion modeling: bridging the gap between farming systems to catchment soil loss in the highlands of Rwanda

Abstract
The current spatially distributed erosion models are often not (yet) capable of continuous simulation of long periods because they do not always include a daily water balance, plant growth and development. This leads to difficulties in incorporating a high degree of spatial information, especially land use information, so that the effects of land use changes on soil erosion have hitherto not been investigated in detail using these models. A new spatially explicit distributed hydrological model was created that couples a spatial daily soil water balance to event based erosion principles with a minimum dataset at high spatial detail. By assuming that the rainfall and overland flow is active during a given number of hours per day, it is possible to calculate rainfall energy and runoff stream power, and use these to simulate sediment detachment, transport and deposition. The streampower principles allow the simulation of both detachment and deposition at a pixel scale (30x30 m) that is in the order of agricultural field size (0.01 – 1.2 ha) in south west Rwanda, while at the same time being able to route landscape sediment transport to catchment outlet. The physical processes implemented in the model are; rainfall, interception, infiltration and vertical movement of water in the soil, overland flow, channel flow, ground water flow, detachment by rainfall and throughfall, detachment by overland flow and transport capacity of the flow. Data inputs to the model include daily rainfall, daily Potential evapotranspiration (ETp), rainfall intensity and spatial maps of: texture class, soil depth, land cover and DEM, and soil and land cover characteristics. The model was tested on the 506 km^2 Rukarara catchment in Rwanda and calibrated using observed daily peak and baseflow. The model predictive capability of reproducing field situation of peakflow, discharge and sediment delivery by surface runoff from hillslopes to river is reasonable with Nash-Sutcliffe efficiency of 68% and 65% respectively for calibration and verification periods (respectively 2010, 2011). The model is sensitive for the infiltration parameters and permeability, soil depth, duration of runoff and rill density that determines erosion intensity. The simulated field level sediment dynamics for different land uses are in line with soil loss found in many field studies in similar environments in Rwanda and the Lake Victoria basin with sediment delivery ratio (SDR) of 27%. The model is therefore a useful tool for soil conservation planning. Immediate improvements can be done in terms of a better representation of crop growth, either by crop growth modelling or for instance NDVI time series. The groundwater dynamics and coupling between the groundwater and baseflow has to be improved but this needs a better process description and also a more detailed hydrological dataset.

Keywords: Distributed models; catchment hydrology; GIS; soil erosion; minimum dataset

4.1 Introduction

Soil erosion is an important phenomenon to understand because it takes away the top soil, removing fertile land from agricultural use. Soil erosion is therefore related to; loss of soil fertility, long-term on-site reduction in crop yield (FAO, 1994; Lal, 1998) and sustainability of agricultural production (Pimentel et al. 1995; Crosson, 1997; Stocking, 2003; Smaling et al., 2012). Erosion adversely affects crop production by reducing soil depth and the availability of water, nutrients, organic matter and as the topsoil thins (OTA, 1982). Potentially negative off-site impacts of soil erosion on water quality are related to sediment and nutrient loading to stream network posing problems for hydraulic infrastructures such as reservoirs, preservation of fluvial ecosystems and eutrophication (Bilotta and Brazier, 2008). Soil erosion problems in Rwanda occur everywhere as a consequence of complex hill-slopes, land use changes, intensive farming with inappropriate agricultural practices on steep slopes that leave the soil disturbed and unprotected (Clay and Lewis, 1996; Musahara and Rao, 2009). Soil erosion rates in Rwanda on cultivated land exceed soil formation rates by more than tenfold e.g., 30 - 150 $tha^{-1}yr^{-1}$ (Konig, 1992; Roose and Ndayizigiye, 1997) and are among the highest in the world (Pimentel et al. 1987, 1995). A recent series of field measurements of soil erosion in the highlands found lower values than reported before that range from 12 t ha^{-1} yr^{-1} to 42 $tha^{-1}yr^{-1}$ on slopes ranging from 25% to greater than 60% (Kagabo et al., 2013). The reported figures are a result of several plot scale studies that are often extrapolated to field and catchment scale (e.g, Lewis, 1988; Konig, 1992; Clay and Lewis, 1996; Roose and Ndayizigiye, 1997; Kagabo et al., 2013). Soil erosion rates in Rwanda are exaggerated because of inappropriate extrapolation from point measurements to field or catchment scale without taking care of sediment sinks and sources. Large scale efforts to estimate erosion rates are missing and general patterns of soil loss remain unstudied in Rwanda. Plot scale studies cannot be scaled up to a catchment size simply on the basis of the catchment size. The sinks and sources of water and sediment make soil loss a very non-linear process in the whole catchment as obstacles (grass strips, hedges on plot boundary, tillage across the slope, contour ridges), and depressions capture sediment locally (e.g., Figure 4.1a and 4.1b). Also, erosion data collection with field plots for large catchment is time consuming and expensive. Normally, erosion models with sediment delivery ratio are used (Smith et al. 1984; Sadeghi et al. 2007), or the detachment and deposition processes are explicitly modeled using distributed models that can take care of sediment sinks and sources using the stream power principle and be able to get a realistic catchment soil loss estimate (Jetten et al, 2003). Bridging the gap between plot and catchment scale measurements using a physically distributed erosion model is important for getting a realistic farming system and catchment soil loss estimates in the highlands of Rwanda.

Figure 4.1a: Field picture in the highlands of south-west Rwanda showing potential areas of sediment deposition by grass strip planted on a contour ridge, trapped under a woodlot plot below

Figure 4.1b: Field picture in the highlands of south-west Rwanda showing smallholder crop plots and potential areas of sediment deposition under grass strips on ridges and a woodlot plot below

Figure 4.1c: Field picture in the highlands of south-west Rwanda showing rill density on smallholder farm plots

There are many physically distributed erosion models for catchment soil loss simulations (Jetten and Favis-Mortlock, 2006). The choice of model depends on the spatial and temporal scale of the processes in the Rwandan highlands. Sediment delivery processes depend on local factors, such as sediment detachment and flow transport travel time, changes in slope, obstacles and depressions that capture sediment locally. Therefore, the model needs to be able to operate on the spatial scale of the farm size that is in order of 0.01 to 1.2 ha in the farming systems of south west Rwanda (Wasige et al. 2013a) and it needs to be able to take the growing seasons into account. For the complex hillscape of Rwanda highlands, you cannot use empirical sediment delivery ratios models of the famous universal soil erosion equation (USLE) type (Wischmeier and Smith, 1978). The USLE model does not calculate soil loss but sediment production only because it has no equations or algorithms to calculate transport and sedimentation processes. Thus it only works in situations for which it was conceived, sloping fields or parts of hillslopes that have little deposition, only erosion. It may only be used on a catchment scale when combined with a catchment based sediment delivery ratio system. It's appropriate to apply a physical distributed erosion model to explicitly account for sediment balance resulting detachment, transport and deposition processes per grid cell at high spatial resolution and connectivity to the river system. Eventually we want to model the effect of land use changes and so the model needs to be able to react to land use changes, so there has to be a close link between the land use and model processes.

4.1 Advances in distributed soil erosion modeling

Advances in sediment modeling have included the development of a grid or cellular approach, dividing the landscape into cells (often square grids) at which basic computations are undertaken and summed for the catchment. For distributed models with in-stream components, the outputs for each cell are then routed through the system to produce catchment scale outputs. This approach subsequently provided a common basis for the structure of distributed hydrologic and water quality models (Moore and Gallant, 1991). They are built on numerical algorithms based on the physics of infiltration, overland flow and channel flow to model the transient response of a catchment. Distributed models are able reflect the spatial variability of processes and outputs in the catchment of analysis. Since the model parameters mostly have a physical meaning, they can be evaluated from direct measurements and without the need for long term hydro-meteorological records (Smith et al. 1995). Jetten et al. (2003) and Merrit et al. (2003) give an extensive description and discussion of these models. The commonly applied distributed models in erosion studies include; AGNPS, EUROSEM, WEPP, LISEM, SWAT and PESERA. In the following section (1.2.1), we review these selected models and also discuss a basis for development of a new spatially explicit distributed model. We have narrowed the range of models to those that explicitly consider sediment and erosion feedbacks. Model types are distinguished in terms of how the physical processes of sediment detachment, transport and deposition are represented by the model, as well as the spatial and temporal resolution of the model types.

4.2.1 A review of commonly used distributed models in erosion studies

The AGNPS model (AGricultural Non-Point Source Pollution Model) (Young et al., 1989) is a distributed event and continuous based model with model parameters for each grid. AGNPS is a tool for use in evaluating the effect of management decisions impacting water, sediment and chemical loadings within a watershed system. AGNPS contains a mix of empirical and physically based components. The model utilizes components of existing models in its structure including the Revised USLE for predicting soil loss in grid cells. AGNPS uses a grid cell representation of the catchment, with cell resolution ranging from 0.4 to 16 hectares (Merrit et al. 2003). Panuska et al. (1991) identified that the grid size selected by the model user was a major factor influencing sediment yield calculations. The European Soil Erosion Model (EUROSEM) model (Morgan, 1994), is a single-event, process-based model for predicting soil erosion by water from fields and small catchments. The representation of a catchment by a cascade of plane and channel elements in the EUROSEM model needs lumping of some parameters for small areas thus averaging out spatial variations of topographical information within such a unit, which represents a drawback when representing large catchments (Saavedra, 2005). The WEPP (water erosion prediction project) watershed erosion model (Lane et al., 1992) operates either a single event or continuous time scale. Despite the process-based nature of the model, WEPP still contains a degree of empiricism and care should be taken when applying the model to new sites. There are a number of possible criticisms of the WEPP model. Firstly, the large computational and data requirements of the model may limit its applicability in catchments where there is often few data or available resources. Many of the model parameters may need to be calibrated against observed data in such studies, creating problems with model identifiability and the physical interpretability of model parameters. The watershed version of WEPP may be of limited applicability to large scale catchments, as simulation involves individual hillslope scale models being 'summed up' to the catchment scale, greatly increasing overall data requirements, model complexity and raising issues of error accumulation (Merrit et al. 2003). LISEM model (the LImburg Soil Erosion Model) is also is a single-event, process-based model for predicting soil erosion by water from fields and small catchments (de Roo and Wesseling, 1996). The LISEM model requires detailed spatially and temporally variable data inputs. The performance of LISEM ultimately is constrained by the resolution and quality of these GIS inputs. As the extent and quality of GIS databases improves, the value of fully distributed models will increase. However, as most other physics-based models, LISEM can be expected to suffer from difficulties associated with identifiability and data availability (Merrit et al. 2003). The Soil and Water Assessment Tool (SWAT) is a semi-distributed conceptual model that works on a daily time step and uses hydrologic response units as the basic computational unit to group input information about land use and soil and management combinations (Neitsch et al., 2002), thus averaging out spatial variations along hillslopes and topographical information within such a unit, but which is essential for sediment generation and transport. In comparison with the grid based erosion models, these semi-distributed models fail at incorporating higher degrees of spatial information, especially the dominant factors affecting soil erosion, such as land use and soil information. Meanwhile, the erosion simulation using daily time steps is based on a modified USLE version which may not capturer soil erosion caused by intensive rainstorms well (Wang et al. 2012). The Pan-European Soil Erosion Risk Assessment (PESERA model) offers a state of the art erosion risk assessment (Kirkby et al, 2000). Erosion predictions from the PESERA model rely on estimating the generation of overland runoff from a daily soil water balance on a cell by cell basis. The model calculates sediment dynamics based on rainfall and runoff energy and uses transport capacity equations to route the sediment to the river system. The model's robustness and flexibility has been demonstrated through its performance at different resolutions and across different agro-ecological zones. Currently it would be possible to use PESERA for this research, but at the start of this research the model was not yet fit for this spatial and temporal scale.

Summarizing, not any of the models above are considered to be suitable for our analysis and it was decided to create a new spatially explicit model with a minimum dataset:

- To model sediment budget at a spatially resolution of 30 x30 m pixel scale which enough for small farmer plots in Rwanda that are in the size of 1 to 2 pixel size,
- To bridge the gap between event based and daily, growing season and annual estimation of soil erosion,
- To connect plot scale erosion to catchment outlet using kinetic energy and the streampower principles that allows the simulation of spatial detachment, transport and deposition important for accounting for sediment sinks and sources and get a realistic catchment soil loss estimates,

With the new spatially explicit model, we should be able to answer the following research objectives:

- Can we produce soil loss information on a small scale field size level with a catchment sized grid based model that requires a minimum dataset?
- How far does the catchment soil loss differ from the soil loss at plot/farm level and where are the sources and sinks of sediment in the catchment?
- Which types of land use produce the most sediment and are these situated on specific slope segments.

This will help in determining the role of land use changes on the sediment and nutrient loads of the river system. Furthermore, the model has to be able to work at relatively high spatial resolution to be able to use LUCC and DEM information derived from satellite imagery, and with a daily temporal resolution to capture growing season changes and enable calibration with daily hydrological measurements. A new model was therefore created that couples a spatial daily soil water balance (including groundwater) to event based erosion principles. The model has one soil layer which is split in an unsaturated zone and a groundwater zone, with a variable boundary between the two compartments that fluctuates with the incoming and outgoing fluxes. The erosion module is implemented by assuming that the rainfall and overland flow is active during a given number of hours per day. This makes it possible to calculate rainfall energy and runoff stream power, and use these to simulate sediment detachment, transport and deposition. There are two advantages to using rainfall energy and streampower principles: i), is the direct coupling with land use, which directly affects infiltration, splash protection and overland flow resistance, and ii) streampower principles allow the simulation of both detachment and deposition to get a realistic catchment soil loss estimate. In this way, we can avoid the simplified use of empirical factors in the modelling such as the C-factor (cover) and P-factor (management) of empirical USLE type models or assumed sediment delivery ratios. The model was tested on the 506 km^2 catchment of the Rukarara river in south west Rwanda, which is dominated by mountains and steep hills, and has seen strong changes in land cover from primary forest to agricultural land use types of; annual crops, Banana, Coffee and Tea in the last 100 years (Wasige et al. 2013b). The combination of steeply graded land, high amount of seasonal rainfall, land use changes and poorly managed soil conservation practices leads to severe soil erosion in the region (Kagera Basin Monograph, 2008; Musahara and Rao, 2009; Bizoza and De Graaff, 2010). The 2010 land cover and one year climate of 2010 to 2011 climate data was applied to simulate soil erosion in spatially explicit quantitative terms (at 30 m x 30 m pixel level). To assess spatial patterns of erosion, soil loss maps were re-classified into erosion classes and compared to land use types and slope classes.

4.3 Materials and methods

4.3.1 Formulation of the spatial distributed soil water balance and erosion model

The spatially distributed hydrological model is a single layer soil water balance with daily timestep. The soil layer contains of both an unsaturated zone and a groundwater zone, with a day to day fluctuation, but it is considered homogeneous in hydrological properties. The process descriptions are borrowed from a number of existing and proven models, based on our state of the art knowledge on hydrology and natural hazards. The principles are a combination of the WOFOST model for the soil water balance (van Diepen et al. 1989), and LISEM for soil erosion and runoff hydrology (Hessel et al. 2003). The soil water balance is translated into a systems approach where we locate all the water storage in a catchment (canopy, soil layer) and the water flows or fluxes between them (infiltration, transpiration, percolation, groundwater flow and runoff). The model output includes all average fluxes in mm/day and cumulative fluxes in mm, spatial output of all fluxes per day and cumulative and outlet discharge, peak and baseflow. The model is written in the scripting language PCRaster (Karssenberg, 2002) and basically any parameter can be written as output.

4.3.2 Model description

The schematic representation of the model is given in figure 4.2. The first thing that happens to rainfall is interception by the canopy of natural vegetation and crops. This water evaporates directly after the rainstorm and is not available for runoff or groundwater recharge. Interception is based on the vegetation canopy storage (Smax, mm) which can be derived from the Leaf Area Index (LAI, m^2/m^2) following De Jong and Jetten (2007);

$$Smax = 0.912 ln(LAI) + 0.703 \tag{4.1}$$

The LAI is derived from the plant cover using the following the equation (WOFOST, van Diepen et al., 1989):

$$LAI = \ln(1-C)/-0.4 \tag{4.2}$$

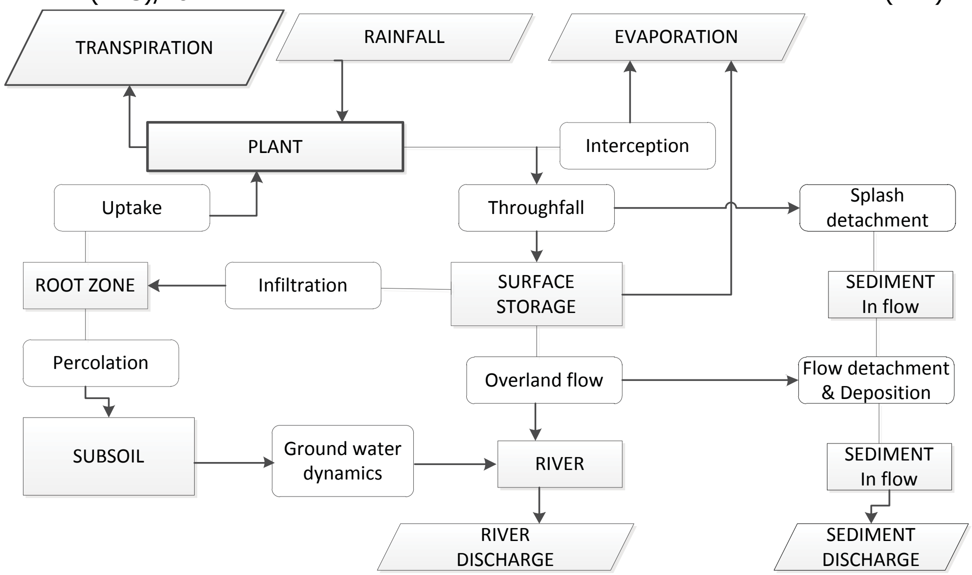

Figure 4.2: Model flow chart

Currently, cover (C) is assumed to be constant for natural vegetation and permanent crops, and for the annual crops, a changing cover is assumed with two growing seasons per year. For each day a canopy water balance is simply the water stored from the previous day, the incoming rainfall and the outgoing evaporation. Potential evapotranspiration, ETp is the water loss that will occur if there is no deficiency of water in the soil for use by the vegetation. ET_0 which is the ETp of reference vegetation, i.e. a uniform short grass cover is used. ET_0 is computed using the FAO-56 Penman Monteith equation (Allen et al. 1998) from the meteo-station parameters. The actual evapotranspiration ETa is smaller than the ETp if the soil is dry. Following the WOFOST model (van Diepen et al. 1989) ETa is assumed to increase linearly from 0 at wilting point to ETp at soil moisture content between field capacity and wilting point.

The infiltration and runoff processes here are simplified to reflect the temporal scale of 1 day. The amount of runoff is determined as the effective rainfall (Pe) multiplied by a runoff fraction *f* that is related to the Ksat ($mm.d^{-1}$) as follows:

$$f = a + b/Ksat \qquad (4.3)$$

a and b were calibrated to 0.3 (-) and 0.58 ($mm.d^{-1}$) to have the best fit between measured and simulated peak flow (see below). The fraction of runoff therefore decreases strongly when the soil becomes more permeable. The water that does not run off is seen as daily surface storage (SS in $mm.d^{-1}$). This surface storage will infiltrate into the soil, but the infiltration cannot be more than the available storage in the soil, determined by the daily unsaturated soil depth (z_u in mm), porosity (θs) and daily moisture content (θ). The infiltration capacity (I_c in $mm.d^{-1}$) is the smallest of the amount of water at the surface and the available storage (SS) in the unsaturated zone:

$$I_c = \min(SS, z_u(\theta s - \theta)/dt) \qquad (4.4)$$

The water available for runoff (RO_t in $mm.d^{-1}$) is thus the runoff fraction multiplied by the sum of the water on the surface from the previous day (SS_{t-1}) and the effective rainfall Pe_t of the current day (rainfall – interception):

$$RO_t = f\,(Pe_t + SS_{t-1}) \qquad (4.5)$$

Where t and t-1 are the current and previous timestep. The new surface storage is then simply calculated by subtracting the runoff. A simple accumulation of RO over a predefined flow network is used as routing mechanism, instead of a true flow algorithm, since we need only the total runoff per day. The flow network is based on the steepest slope in eight directions. The accumulated runoff reaches the river channel and becomes the peak flow component of the discharge.

Finally, percolation is calculated as the water that flows from the unsaturated zone into the groundwater layer. Percolation depends on soil characteristics and soil moisture conditions. Normally in the soil, water moves under the influence of gravity and differences in suction (capillary forces). The percolation flux equals the unsaturated conductivity, $K(\theta_E)$ (in $m.s^{-1}$) assuming that there is only gravity to move the water. We use the equation of Van Genuchten (1980);

$$K(\theta_E) = Ksat * f(\theta_E) = \mathrm{Ksat}\sqrt{\theta_E}\left[1-(1-\theta_E{}^{\frac{1}{m}})^{\mathrm{m}}\right]^2 \quad (4.6)$$

where; m is a texture dependent parameter with guideline values from Van Genuchten *n* parameter (Hodnett and Tomasella, 2002);

m=1-1/n (4.7)

Combining all the fluxes, the soil water balance is presented as:
dS/dt = P- (Interception-Actual evapotranspiration-Percolation-Runoff) (4.8)

where; *dS* is the difference in soil moisture (in mm) at the beginning and the end per time step, and dt is the time step (1 day).

Note that the unsaturated zone depth changes per day as the groundwater level rises or falls. However it cannot become smaller than a fixed "margin" of 300 mm (calibrated) to have always some soil water buffer.

We model groundwater and subsurface flow leading to base flow. The ground water balance is implemented in the same conceptual way as the soil water balance: by calculating incoming and outgoing fluxes for each cell. First we assume an initial groundwater level at the start. Then for each timestep we add the incoming flux (percolation from the soil layer above) and subtract the outgoing fluxes (flow to neighbouring cells). The hydraulic potential *H* is the sum of the groundwater level *h* and the absolute elevation at a location *z* (*H*=*h*+*z*). Below the groundwater level the soil is fully saturated, so we can use the saturated hydraulic conductivity, *Ksat*. When there is a difference in hydraulic potential *dH* between two points (over a distance *dL*) water will flow from the higher to the lower potential;

$$Q_{GW} = qA = Ksat\left(\frac{dH}{dL}\right)h\,dx = Ksat\left(\frac{dh+dz}{dL}\right)h\,dx \quad (4.9)$$

Where; Q_{GW} is the groundwater flow in m^3s^{-1}, *q* is the one dimensional flux in m/s and *A* is the cross section of flow (m^2), which is the product of the cell width *dx* and the water height *h*. *Ksat* is the saturated hydraulic conductivity (ms^{-1}). This equation is solved in 4 directions in the gridcell system, and the resulting groundwater flow is the sum of the 4 fluxes (NS and EW). The ground water is drained by the river system, and becomes the baseflow component of the discharge. Groundwater enters the stream channel from the side and from below. In the model, all groundwater over the width of the stream channel becomes baseflow. For our study area, it is assumed that all baseflow in the stream channel is transported to the outlet in less than 1 day (the measurements show that generally the response time is less than 12 hrs).

4.3.3 Soil erosion module

The erosion modelling consists of equations to generate sediment with splash detachment (Ds) and flow detachment (Df) and to move sediment in the landscape using transport capacity (TC). The equations used here are derived from the LISEM model (Baartman et al., 2012) and normally used for event based erosion calculations and not suited for daily time-steps. We therefore have to assume a period in the day of "rainfall and runoff activity". The assumption is that all activity has duration of 1 hour. This allows us to calculate the rainfall intensity needed for rainfall energy, and the flow velocity needed for flow energy (stream power). The concept is the same for splash and flow detachment: they are calculated as the product of soil strength and the energy needed to dislodge particles. The splash detachment is determined by the kinetic energy of both direct through fall and from drainage from leaves. Rainfall kinetic energy (KE_{dr} in J.m^{-2}) is calculated from the rainfall intensity and amount in the following equation (Van Dijk, 2002):

$$KE_{dr} = KE_{max}(1 - 0.52\exp(-0.042I)) \tag{4.10}$$

Where KEmax = 28.3 Jm-^{2}, intensity I (in mm h^{-1}) is the daily rainfall divided by the average duration of 1 hour. KE_{dr} is multiplied by the rainfall directly reaching the soil (P in mm) and not intercepted by the canopy to obtain the daily KE (Jm^{-2}day^{-1}) (see equation 12 below).

The water dripping from the leaves after the interception storage is filled up, also contributes to the splash detachment. The kinetic energy of this canopy drainage flux is calculated differently, because the fall velocity depends on the plant height. The empirical equation of Brandt (1990) is based on plant height (PH in m):

$$KE_{ld} = 15.8\sqrt{PH} - 5.89 \tag{4.11}$$

Where; KE_{ld} is the kinetic energy of leaf drip in Jm^{-2}. Note that *PH* must be larger than 0.14 m in order for the equation to work. Combining equation 10 and 11, we get the total kinetic energy on a day (KE) (in Jm^{-2}) calculated as:

$$KE = CPe\,KE_{ld} + (1 - C)P\,KE_{dr} \tag{4.12}$$

where, C is the canopy cover fraction, *Pe* is the effective rainfall (mm) and *P* is the direct rainfall (mm), KE_{ld} is the kinetic energy of leaf drip in Jmm^{-1}m^{-2} and KE_{dr} is rainfall kinetic energy (Jm^{-2}).
The splash detachment, Ds (in kgm^{2}) is calculated as:

$$Ds = K_{sp}KE \tag{4.13}$$

Where; the erodibility K_{sp} is an empirical parameter in kg soil loss J^{-1} rainfall energy (Morgan et al. 1998).

The flow detachment and transport capacity depends on the unit stream power, which is a measure of the energy of the overland flow. The transport capacity of overland flow is modelled as a function of unit stream power (after Govers, 1990), which we use for an average texture grain size:

$$TC = 2650 \, * \, 0.91 \, (VS)^{0.65} Q \tag{4.14}$$

Where; TC is the gridcell transport capacity (in kgs^{-1}),*V* is the overland flow velocity (m/s) and S is the tangent of the terrain slope, discharge Q (in m^3s^{-1}) is the product of the flow velocity with the wet cross section *A* (in m^2) and 2650 kgm^{-3} is the average particle density of sand. The overland flow velocity is needed in ms^{-1} and is derived from the daily runoff amount RO (mm), whereby it is assumed that the runoff is active during one hour. This allows us to calculate an average runoff water height (h_{ro}) per second that is used to calculate the velocity. We also assume that the water does not flow as sheet flow over the entire gridcell but is concentrated in rills (see Figure 4.1c), using a rill density factor of 0.05 (of the grid cell):

$$h_{ro} = (RO/3600)/\text{rill density} \tag{4.15}$$

The velocity is calculated from the Mannings equation:

$$V = h_{ro}^{2/3} \sqrt{(S)/n} \tag{4.16}$$

Where S is the terrain slope, n is the Manning's flow resistance factor (based on land use type), calculated based on the ground cover (GC given as input), and based on Hessel et al. (2003):

$$n = 0.01 + 0.1*GC \tag{4.17}$$

The equation for detachment of particles from the soil surface and for the transport of particles is the same, except that the soil cohesion must be overcome to dislodge the particles. This is expressed in an empirical detachment efficiency (Y, dimensionless) related to cohesion (Morgan et al. 1998):

$$Y = 0.79 \, e^{-0.85coh} \tag{4.18}$$

Where; *coh* is the sum of the soil cohesion and plant root cohesion (in kPa). Guideline values for soil cohesion for different soil types are given in Morgan et al. (1998). With this parameter, the equation for flow detachment, Df (in kgs^{-1}) is:

$$Df = Y\,TC \tag{4.19}$$

Whenever the transporting capacity is less than the total available sediment from splash, from up-slope areas and from previous time steps, deposition occurs. If the transporting capacity of the flow exceeds the available sediment, detachment takes places.

4.4 Description of the study area

The study site is the 506 km^2 river Rukarara catchment, is a tributary of the Kagera river in the hilly upper reaches of lake Victoria basin (LVB) ($2^0 28'$-$2^0 34'$S and 29^0 $23'$-$29^0 29'$E) where forest degradation and inappropriate agricultural practices by smallholder farmers are causing the region's topsoil to erode (Kagera Basin Monograph, 2008; Musahara and Rao, 2009). The steep hillslopes exhibit variability in terms of elevation, climate, geology, and physical soil properties as the river Rukarara drains and descents the catchment. The river has its source at about 3000 m.a.s.l. and descends to 1470 m.a.s.l at the outlet (Figure 4.3). The low-land is hotter and drier than the upstream area. Mean annual temperature is about 15^0C - 18^0C in the highland and 26^0C in the low land. The rain pattern is bi-modal, with long rains occurring during September to January, and shorter rains in March to June. Annual precipitation ranges from over 1200mm to 1800mm with a mean of 1500mm. Rain falls mainly during storms producing large amounts of runoff that consequently often acts as a driver of soil erosion (Kagera Basin Monograph, 2008; Musahara and Rao, 2009; Bizoza and De Graaff, 2010; Bizoza, 2011). The geology of the area consists of bedrock formed in the middle Precambrian period and is differentiated in two parts of the catchment. In the eastern part of the catchment, the substratum is derived from granites and granitic rock. In the western part of the catchment, schist, quartzite and dolerite are the main parent rocks. A soil map (1:50000; FAO/Unesco, 1988) of the area is available (Figure 4.4a and b) and reveals that the soils are variable; Acrisols, Alisols, Ferralsols, Gleysols, Histosols and Cambisols occur. The vegetation consists of a forested upper part of Nyungwe natural forest responding to natural, degraded and a buffer strip of plantion forest (Figure 4.5) while the middle to lower parts of the catchment has largely agricultural crops fields. Food crops are typically grown on plots of 0.1 ha or less. Cultivation and tillage by smallerholder farmers is mainly undertaken with the use of simple hand implements (Figure 4.1a and 4.1b).

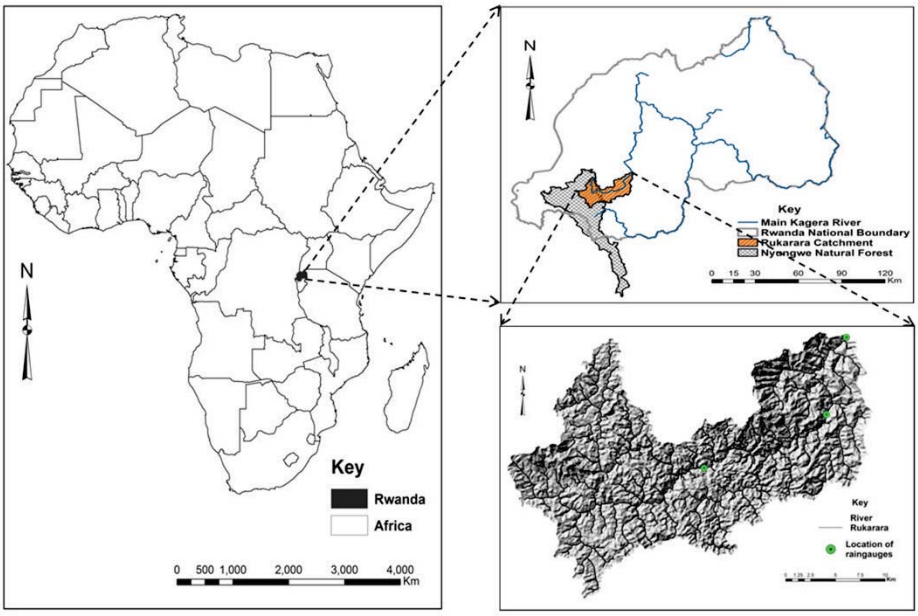

Figure 4.3: Location of Rwanda and hillshade map of Rukarara catchment

4.5 Spatial datasets

Spatial datasets used in the modeling included: a digital elevation model derived from ASTER and maps of land cover, soil types, and tables of soil and land use information. Soil information included (Table 4.1); saturated hydraulic conductivity (Ksat), porosity, field capacity, wilting point and a Van Genuchten parameter for unsaturated flow based on the texture class (see equation 7). Land use information included (Table 4.2); crop height (m), cover fraction, and near surface ground cover. The DEM was an ASTER DEM (30m x 30m). The soil classes (Figure 4.4b) and depth map (Figure 4.6) were derived from a soil map (1:50000 scale of FAO/UNESCO, 1988 classification) of the area developed by (Verdoodt and Van Ranst 2006). On the basis of soil and land cover maps, a total of 178 locations were visited for sampling and measurement at each location during the month of February to March of 2010. The following parameters were measured using standard laboratory procedures described in Okalebo, et al. (2002): soil information of Ksat, bulk density and soil texture. Based on your assessment of the texture, the following soil parameters; porosity, field capacity, and wilting point were derived from USDA textural triangle. Vegetation information collected from the field during field visit included: vegetation height (m) and vegetation cover fraction. Vegetation height was measured using measuring tape to quantify the average height of five plants of each species per measurement site. Vegetation cover (fraction) was estimated from vegetated cover within a $1m^{-2}$ grid: average of three estimations per measurement site. Vegetation parameters for the dense forest and plantation cover were obtained from earlier studies in the same area by Nsabimana, (2009). Land use/land cover maps for the period of 2010 derived from Landsat imagery (Figure 4.5) was already available from Wasige et al. (2013b). Finally the soil depth (see figure 4.6), which is linked to the total hydrological storage in the soil profile for soil and groundwater, was estimated using a multiple regression equation derived from Kuriakose et al., (2009), using slope angle, curvature, distance from the river and the original soil map information. This resulted in a map that has deeper soil in the valleys (up to 2 m) and shallower soils on the steeper slopes (up to 0.5 m). Subsequently the soil depth is included in the calibration procedure to create more or less storage space.

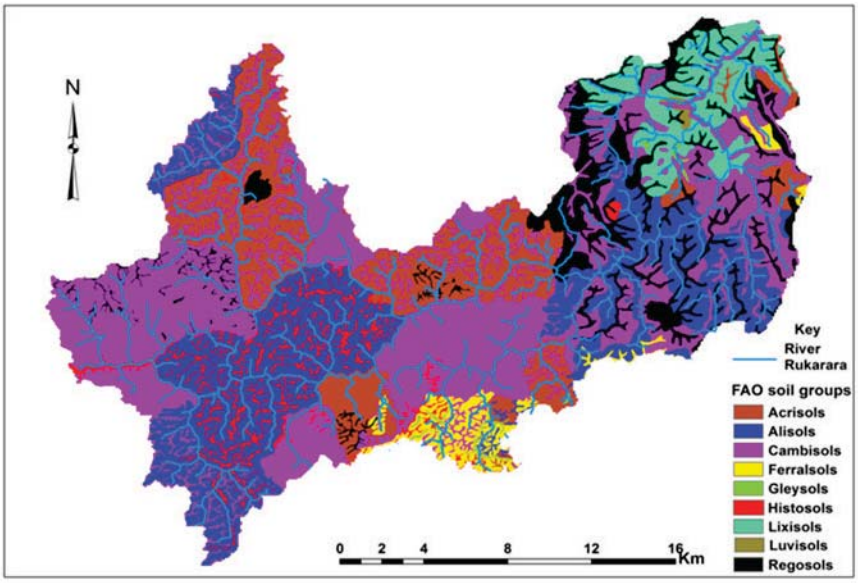

Figure 4.4a: Dominant FAO 1988 soil groups

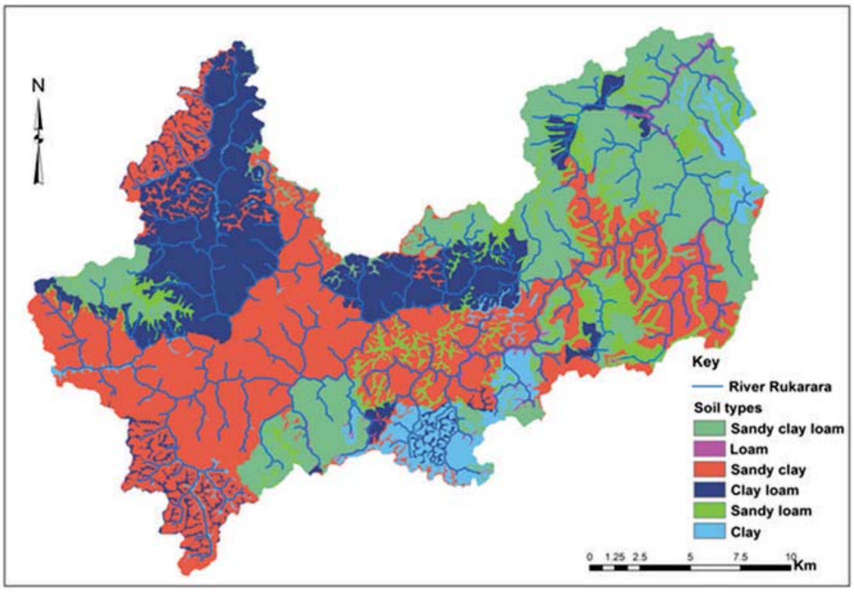

Figure 4.4b: Texture class map of Rukarara catchment

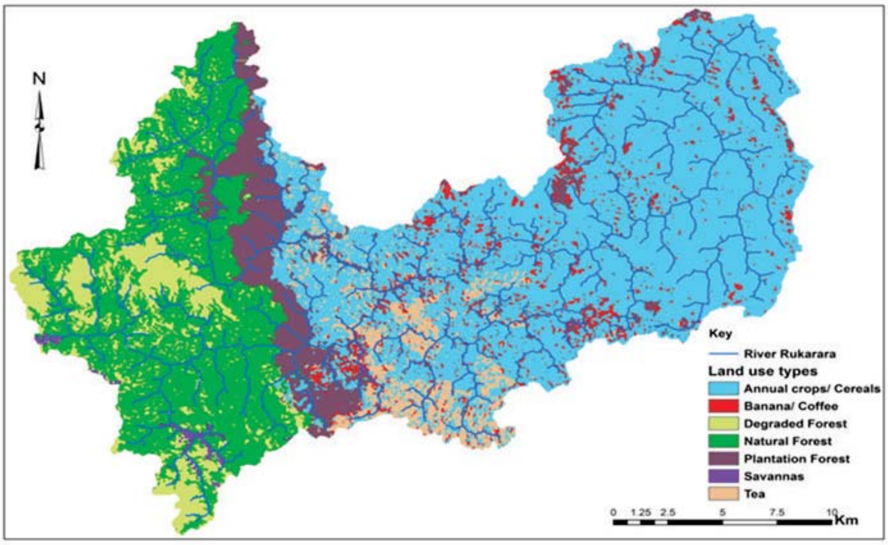

Figure 4.5: Land cover types of Rukarara catchment during 2010

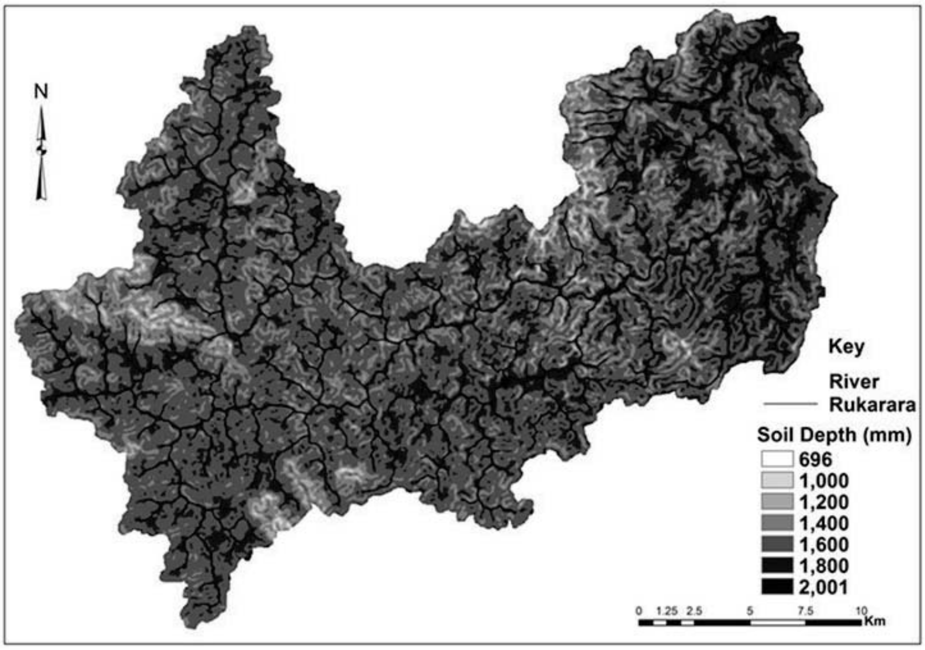

Figure 4.6: Soil depth map in Rukarara catchment

Table 4.1: Soil information of Rukarara catchment used on the modeling

Soil types	ksat ($mm.h^{-1}$)	Pore (multiple factor)	Field capacity (multiple factor)	Wilting point (multiple factor)	van genuchten n	soil strength parameters
Clay	8	0.52	0.4	0.29	0.34	0.47
Clay loam	12	0.51	0.36	0.22	0.30	0.67
loam	39	0.46	0.31	0.15	0.31	0.84
Sandy clay	5	0.46	0.38	0.27	0.28	0.84
Sandy clay loam	12	0.43	0.28	0.18	0.35	0.84
Sandy loam	71	0.43	0.22	0.11	0.36	0.84

Table 4.2: Land use information of Rukarara catchment used in hydrological modelling

Land use types	Cover (%)	Plant height (m)	Ground cover (%)	manning's n
Annual crops	0.6	1	0.1	0.04
Banana-Coffee	0.7	3	0.2	0.08
Degraded Forest	0.8	8	0.4	0.15
Natural Forest	0.9	15	0.6	0.12
Plantation Forest	0.9	15	0.5	0.08
Savannas	0.9	1.2	0.7	0.15
Tea	0.8	0.5	0.2	0.06

4.6 Temporal dataset

Rainfall

For our study, we installed and daily monitored three manual rain gauges (CASELLA M114003,Splayed base) strategically placed in open areas and at about 30cm from the ground at 3 points along the river gradient representing the upper forest area, middle and lower (outlet) agricultural area (Figure 4.3). According to Bruce and Clark (1966), for comparison of seasonal precipitation figures with seasonal runoff volumes, stations as few as 'one per 500–750 km^2 usually suffices. The approach of weighted distance was used to aggregate the rainfall to the discharge at the outlet (Figure 4.7a). A Vantage Pro2 automated weather manufactured by Davis Instruments Corp was also installed in upper forest area of the study site to record (hourly) extra data on rainfall, rainfall intensities, maximum and minimum temperature.

Water level measurement and Discharge calculation

A stage gauge was installed and water levels were recorded twice daily (7:00 and 19:00 Hrs) and immediately after every storm during a 3 year period (1st march 2010 to 29th February 2013). Sediment samples were collected once a week. For determination of water velocity, at least 15 readings were taken during a low, high and no (dry) rainfall events. Cross-sectional area of the river was measured twice a year and the average determined. River discharge, Q (m^{3}s^{-1}) was determined from water level and velocity data using a site-specific stage-discharge relationship (Figure 4.8) and discharge computed as;

$$Q=AV \tag{4.20}$$

where; A is area (m^2) and V is velocity (ms^{-1}).

We have measured the discharge (see Figure 4.9) and can separate this in a peakflow and baseflow component using a simple linear increase method which assumes a slight linear increase of baseflow from the start of a peak event to the end (equation 21).

$$Qb_t = Q_{t-1} + dQb_t \tag{4.21}$$

Where Q_{t-1} is the discharge at a day (m^3.s^{-1}) when there is no rainfall. The increase in baseflow dQb is empirical and set at 0.5 m^3.s^{-1}. This value gave the best distinction of individual peak flows. The increase stops when the baseflow exceeds the measured discharge. The baseflow is then equal to the discharge until the next rainfall. Figure 4.10a shows an example of this method for dry and rainy period.

The baseflow data allowed us to calibrate the groundwater flow, while the peakflow allowed us to calibrate the accumulated runoff. While the discharge was measured at three locations only the outlet could be used for calibration due to time constraints and spatial scale issues that required extra data points on rainfall variability to improve on run-off modeling. Using the measured discharge and peak flow analysis, we use these values to calibrate the simulated ground water movement and surface runoff, and choose the Ksat in such a way that the simulated (see Figure 4.11 for peak flow and Figure 4.12 discharge) and the detailed fluctuations values correspond as much as possible to the measured values. The discharge and runoff are calibrated on a temporally varying signal so that the simulated annual total discharge and peak flow resembles the total observed (m^3yr^{-1}).

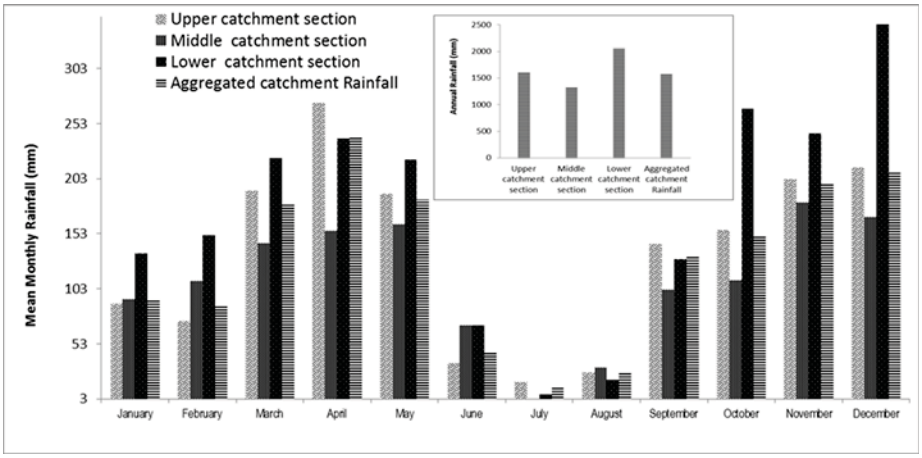

Figure 4.7a: Mean measured monthly and annual (inset) rainfall over 3 year period (March - 2010 to February 2013)

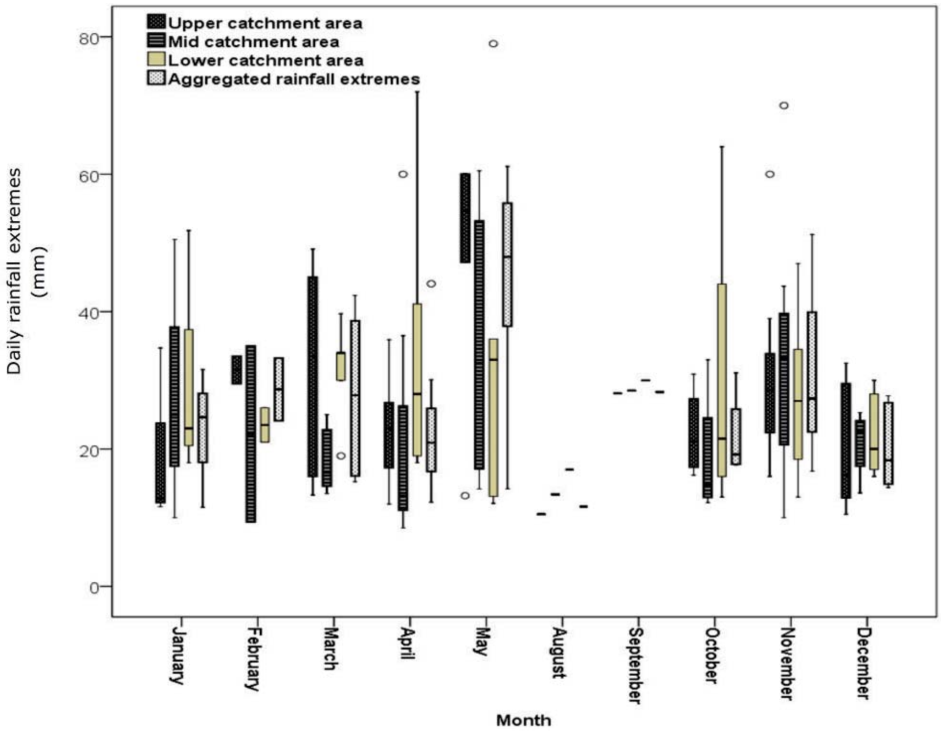

Figure 4.7b: Variability of daily extremes rainfall (above mean daily rainfall) (March-2010 to February 2013). The box-and-whisker diagrams include: median value; (large rectangle) range of 50% of the samples; (cross bars)

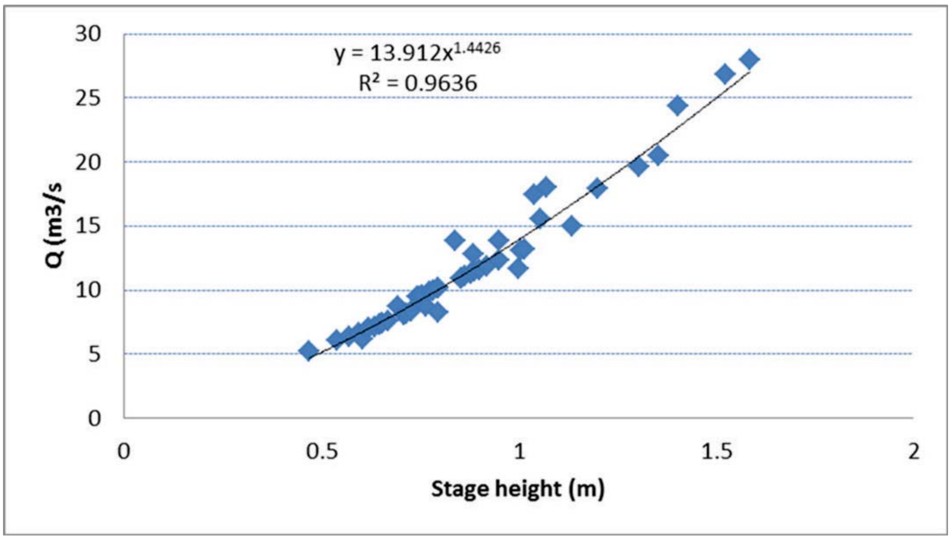

Figure 4.8: Stage-discharge calibration curve for the Rukarara catchment, outlet station

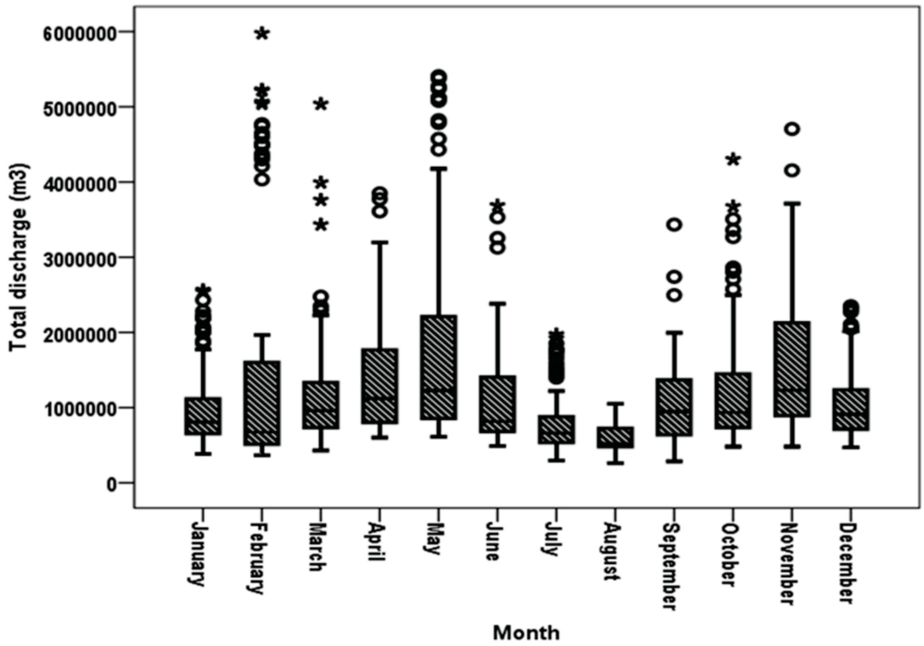

Figure 4.9: Variability of mean monthly discharge over 3 years (March-2010 to February 2013). The box-and-whisker diagrams include: median value; (large rectangle) range of 50% of the samples; (cross bars)

4.7 Model calibration and validation

The model was tested to compare the simulated results to measurements of hydrology. There are two tests that were conducted as the first principles of modeling; (1) mass balance check, and (2) calibration and validation. The mass balance check included testing for the reconciliation of the model soil water balance (equation 4.8).

The parameters subjected to calibration were divided into two groups responsible for runoff and erosion simulations. The parameters responsible for runoff simulation were calibrated firstly using the observed discharge data, which was split into baseflow and peakflow (Figure 4.10b). Table 4.3 lists the parameters calibrated for the rainfall runoff module and soil erosion module, which shows the range and final values resulting from the calibration. Daily accumulated runoff was compared to peak flow and daily accumulated groundwater flow to baseflow. The runoff coefficient parameters determined the daily infiltration and runoff response (higher gives more runoff and less infiltration), the unsaturated soil depth "margin" and total soil depth correspond to water storage in the soil, groundwater Ksat was increased to increase the groundwater output. The "rill density" was used to assume runoff to be concentrated on a small part of the 30m gridcell in rills (a lower density concentrates the water more and increases erosion), and finally the "duration of runoff" determines the velocity (shorter duration increases the velocity and increases transport capacity and detachment).

Table 4.3: Calibration parameters in the model with the final value selected and the range that was tried between brackets.

Variable	**value (range)**
Runoff fraction parameter a (equation 4)	0.30 (0-1) fraction
Runoff fraction parameter b (equation 4)	0.58 (0-1) multiplication factor
Soil depth	2.2 (1-5) multiplication factor
Groundwater Saturated hydraulic conductivity K_{Sat}	5 (1-10) multiplication factor
Minimum unsaturated top soil depth (margin)	300 (150-500) mm
Duration of runoff activity	1 (1-4) hours
Rill density	0.05 (0.05 - 1) fraction

The model was calibrated and checked for discharge, peakflow and soil loss. For the model evaluations, flow datasets were split into a 'training set' for calibration (March 2010 to February 2011 representing the first year) and a 'testing set' for validation (March 2011 to February, 2012 for second the year). Testing of model performance was conducted using the weekly data. Weekly data was preferred because it smoothened the data, creating a better match between the observed and simulated values compared to the daily values. Calibration and validation results were evaluated by statistical comparison of measured discharge. Performance indicators such as Nash–Sutcliffe model efficiency coefficient (Nash and Sutcliffe, 1970) and coefficient of determination (R^2) were used. We did not calibrate the sediment yield module because the available data was collected during the low flows and did not capture high flows. The data represent mostly background sediment values in the baseflow (which the model cannot predict). The only verification that could be done was a comparison of average simulated TSS (total suspended solids), SDR (sediment delivery ratio) and field soil loss values for land use types with literature values of studies reported in Rwanda and the Lake Victoria basin.

Once the model was calibrated and verified, the 2010 catchment condition as the baseline and one year climate of 2010 to 2011 climate data was applied for simulation of the effect of land use types on soil loss. For the field net erosion simulation, rill density and plant cover factors were also subjected to calibration fit to the findings of field studies by Lewis, (1988) and Kagabo et al (2013) on soil loss in Rwanda. To assess spatial patterns of erosion, soil loss maps were re-classified into five erosion classes (see Table 4.4). We had 7 land use types that occur in any slope position. It is likely that they are difficult to be compared because of this, all erosion levels occur in all land use types. Therefore, there was a further subdivision in slope class under the assumption that this would allow a better comparison. The following slope steepness categories; foot slope (0 – 15%), mid slope (15 – 35 %) and upper slope (greater than 35%). The upper slope position includes upper and lower interfluves and receives little or no runoff, but contributes runoff to lower slope positions. The middle slope position includes shoulder, upper and lower linear slopes, and receives runoff from the upper slope and contributes runoff to foot slope positions. The foot slope represents the base of the hill (Wang et al., 2001; Kagabo et al., 2013). Slope positions were derived from an ASTER DEM (30 m x 30 m). Spatial analysis was then used to identify and prioritize erosion-hotspot areas of the catchments that demand appropriate management strategies. Tolerable soil loss was fixed at 10 t ha^{-1} yr^{-1} for tropical regions according to Morgan (1986).

Table 4.4: Categories of soil loss classes (FAO, 1990)

Soil loss by degree	Nature	Threshold (t ha^{-1} yr^{-1})
Very low	slight	0-2
low	low	2-5
Moderate	Tolerable	5-10
High	Bright spot	10-20
Severe	Hotspot	20-50
Very severe	Extreme	>50

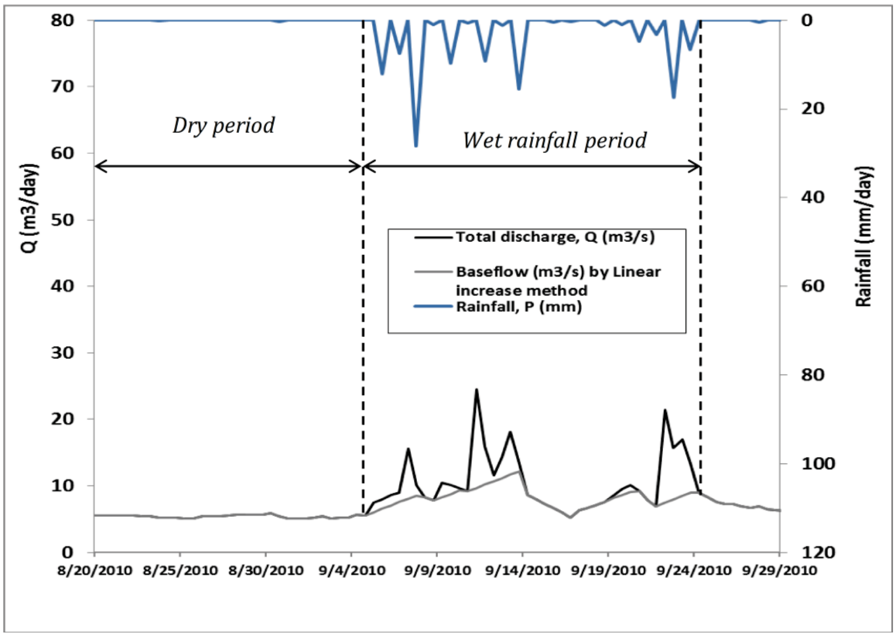

Figure 4.10a: Example of base flow separated from the discharge by simple linear increase (equation 21) during the dry and rainy periods of Rukarara catchment, south west Rwanda

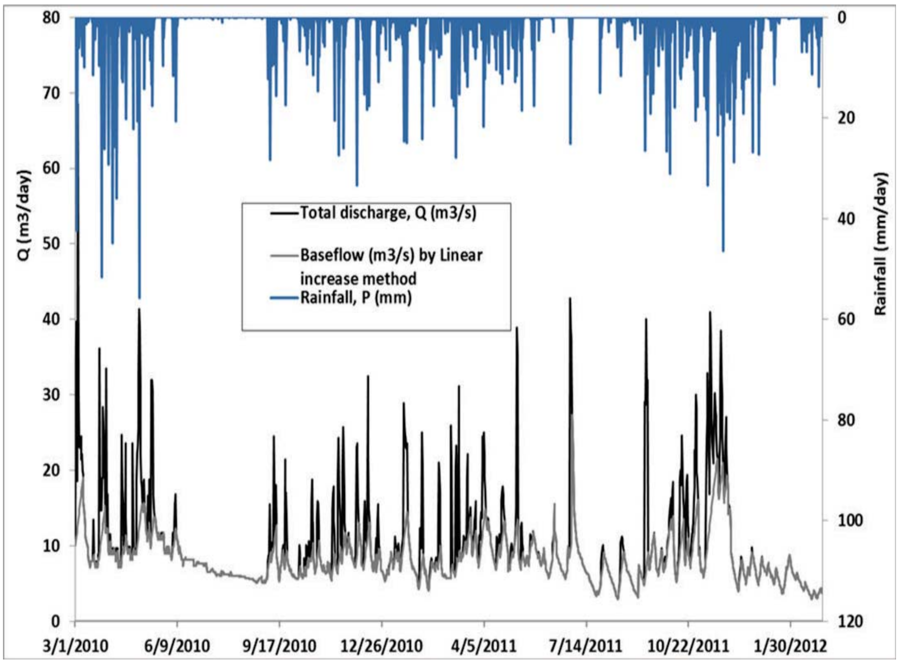

Figure 4.11: A comparison calculated and simulated daily peak flow of Rukarara catchment, south west Rwanda (March, 2010 - February, 2012)

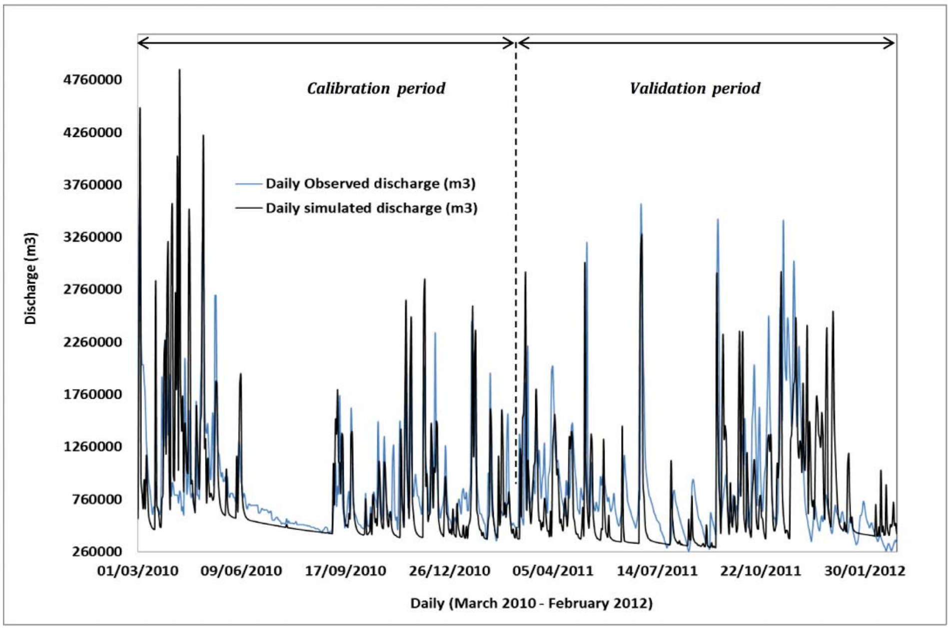

Figure 4.12: A comparison of calculated and simulated daily discharge of Rukarara catchment, south west Rwanda (March, 2010 - February, 2012)

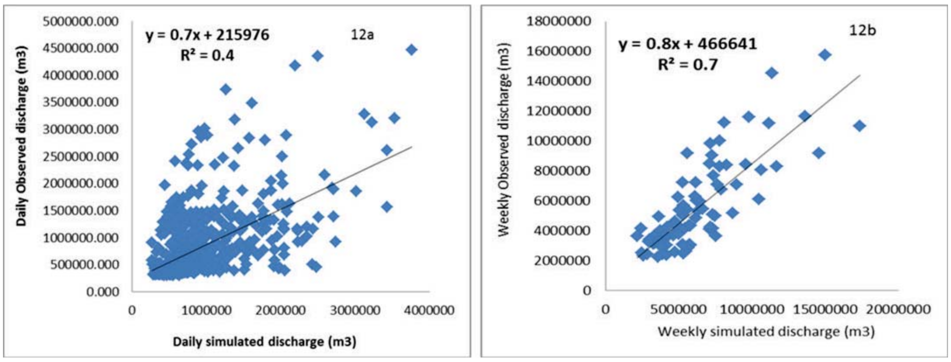

Figure 4.13: A comparison of daily (a) and weekly (b) simulated discharge for Rukarara catchment (March, 2010 - March, 2012)

4.8 Results

4.9 Hydrology

4.9.1 Results of measured rainfall and stream flow

Rainfall is an important physical parameter affecting soil erosion in Rwanda, and its characteristics are essential for understanding runoff and soil loss. Monthly and annual rainfall data during a 3 year study period is presented in figure 4.7a. The rainfall amount of 1560 mm during the period March 2010–February 2013 was of the same range of magnitude as the mean long-term value (1500 mm) (inset Figure 4.7a). Rainfall is bimodal with high amount of rainfall registered through the year apart from the dry months of June, July and august that had a few showers. Over the 3 years period of the study, rainfall extremes occurred throughout the year (6b). Monthly stream flows recorded at the outlet over 3 year period are presented in figure 4.9. Low flows were recorded during the dry months of June, July and august and high flows recorded in May and November.

4.9.2 Calibration and validation results

The results of base flow separation are presented in Figure 4.10b. Outlet-based predictions are presented as total and peak discharge. Using a simple linear separation method, peakflow constituted 19 % of the total discharge. Figure 4.11 and 4.12 shows the observed and simulated peakflow and discharge for the simulation period. The model performs reasonably well when looking at the water balance check (Table 4.5). Using the calibration parameters, the annual totals are simulated with a 1.7% water balance error (24.6 mm error of 1428 mm annual rainfall). This is mostly due to the simplified groundwater movement, based on the explicit groundwater flow equation, which produces errors during periods of strong fluctuations. Looking more in detail to the results, the model performs reasonably well on a level of weekly aggregated values (Figure 4.13b, right) while the daily discharge is not simulated very well (Figure 4.13a left). This is probably due to scale issues: in some cases, the measured data shows peakflow when there is no rainfall registered in any of the 3 raingauges, or shows no peak when there are substantial rainstorms. The model almost always produces runoff when there is rainfall. The three raingauge points were not enough to capture spatial rainfall distribution to correctly match rainfall-discharge relationship. We can therefore conclude that spatial rainfall data, runoff and infiltration description could be improved, although the timestep of one day prohibits the simulation of the infiltration process in detail. Also the valley floor with groundwater tends to fill up and then remains "full", which results in little groundwater movement because there are no pressure differences. It is not known to what extent this reflects reality, although the stagnating groundwater has sustained rice production throughout the year in some flat valley bottoms in Rukarara catchment. The general evolution of the observed hydrograph is replicated rather well (Figure 4.11 and 4.12). Overall, there were more overestimates than underestimates, particularly of larger events. The different performance criteria for the assessment of the model performance are given in table 4.6. Accordingly, the calibrated model is able to reproduce the observed hydrograph with Nash-Sutcliffe efficiency of 68% and 65% respectively for calibration and verification periods.

Table 4.5: Annual water balance check of Rukarara catchment, south west Rwanda (2010/2011). Baseflow and runoff follow from the baseflow separation of the discharge

Water balance parameter	Simulated (mm)	Measured (mm)
Rainfall (P)	*1428.6*	*1428.6*
Actual Evapotranspiration (ETa) + Interception (Intc)	876.7+132.3 = 1009	
Potential evaporation (ETp)		*1158*
Runoff (RO)	124.6	*122*
Soil water balance: P – (ETa + RO + Intc + Perc)	-183.4	
Change in soil moisture (end - initial)	-186.6	
Percolation (Perc)	478.0	
Baseflow	518.6	*512*
GW balance: Perc – Baseflow	-39.4	
Change in GW storage (end-initial)	-61.7	
Mass balance error	24.3	(=1.7% of rainfall)

Table 4.6: Model evaluation criteria and performance results for the calibrated model

Criteria	Reference	Description	Year 1 (Calibration indicator) (%)	Year 2 (Validation indicator) (%)
$1-\frac{\sum(Q_s-Q_o)^2}{\sum(Q_o-\bar{Q}o)^2}$	Nash and Sutcliffe, 1970	Ability of the model to reproduce the time evolution of all discharges.	68	65
R^2	Coefficient of Determination		69	62

4.10 Soil erosion modeling

4.10.1 Soil loss and deposition at field scale

Table 4.7a and Figure 4.14a and b present the nature of soil loss and deposition at field scale. Frequency analysis on soil loss at 30m x 30m pixel for the year 2010 (Table 4.7a) shows that highly eroded areas in the landscape mainly occur in the region with agricultural land cover types (Figure 4.14a). It can be noticed that most of the simulated soil loss varied between 0 – 70 t ha^{-1} yr^{-1}. Few pixels (2 % to 6 %) had soil loss greater than 100 t ha^{-1} yr^{-1}. The class of less than 5 t ha^{-1} yr^{-1} of soil loss represented the largest proportion for all land cover types. The lowest number of pixels with tolerable soil loss occurred under agricultural land uses (annual crop, banana-coffee and tea) indicating high soil loss for this land cover types. Forests are the biggest depositional area (Figure 4.14b).Nearly a quarter of pixels under annual crop cover were soil loss hot spot areas (>20 t ha^{-1} yr^{-1}) and were about 15 % for tea and banana-coffee. Overall, the largest percentage of pixels was under tolerable soil loss (< 10 t ha^{-1} yr^{-1}) ranging between 60% to 93% of the whole catchment (Table 4.7b). About 3 % to 17% of pixels were hotspot for soil loss (> 20 t ha^{-1} yr^{-1}).

Table 4.7a: Percentage (%) of pixel by each soil loss class within land cover type in Rukarara catchment in 2010

Soil loss classes (t/ha)	Annual crops	Banana-Coffee	Tea	Degraded Forest	Natural Forest	Plantation Forest	Savannas
	Percentage (%)						
5	41	52	53	71	81	82	88
10	19	20	17	13	9	8	5
15	11	8	8	5	3	3	2
20	6	4	4	3	1	1	1
25	4	3	3	1	1	1	1
30	3	2	2	1	1	0	0
35	2	1	2	1	0	0	0
40	2	1	1	0	0	0	0
45	1	1	1	0	0	0	0
50	1	1	1	0	0	0	0
55	1	0	1	0	0	0	0
60	1	0	0	0	0	0	0
65	1	0	0	0	0	0	0
70	1	0	0	0	0	0	0
75	0	0	0	0	0	0	0
80	0	0	0	0	0	0	0
85	0	0	0	0	0	0	0
90	0	0	0	0	0	0	0
95	0	0	0	0	0	0	0
>100	6	4	5	2	2	2	2
Percentage of pixels under tolerable soil loss (< 10 t ha^{-1} yr^{-1})	60	72	70	84	90	90	93
Percentage of pixels under Moderate soil loss (> 10 $\leq$ 20 t ha-1 yr-1)	17	12	12	8	4	4	3
Percentage of pixels which are hotspot for soil loss (> 20 t ha^{-1} yr^{-1})	23	13	16	5	4	3	3

Table 4.7b: Percentage (%) of area covered by each soil loss class by land cover type in Rukarara catchment in 2010

Soil class (tons)	Annual crops	Banana-Coffee	Tea	Degraded Forest	Natural Forest	Plantation Forest	Savannas	Total % of the soil class
	% of the number 30 m x 30 m pixels in the soil class							
5	34	4	5	9	34	14	1	100
10	62	6	6	7	14	5	0	100
15	71	5	6	5	9	4	0	100
20	74	4	6	5	7	3	0	100
25	75	4	7	4	7	3	0	100
30	76	4	7	4	7	3	0	100
35	75	4	7	3	8	3	0	100
40	75	4	7	4	7	2	0	100
45	77	4	6	4	8	2	0	100
50	76	4	6	4	8	2	0	100
55	76	4	6	3	9	3	0	100
60	78	4	5	3	8	3	0	100
65	75	4	6	3	9	3	0	100
70	75	4	5	3	9	3	0	100
75	77	5	5	3	7	3	0	100
80	73	4	5	3	10	4	0	100
85	77	4	4	3	9	2	0	100
90	73	5	5	4	11	3	0	100
95	75	3	5	4	10	3	0	100
>100	66	5	6	5	14	4	0	100
% of the overall total	49	4	5	8	24	10	0	100

Mean soil loss predicted by land use type and slope position are presented in Table 4.8. The pattern of soil loss by slope was important for annual crop land use types and was highest in the midslope. The rest of the other land use types had marginal differences in soil loss by slope position. This concurs with earlier findings on SOC content on midslopes under annual crops (Wasige et al. 2013c). The spatial distribution patterns of soil erosion (Figure 4.14a) and deposition (Figure 4.14b) were found to be closely related to the topographic and landscape features in most cases. Soil loss was high in the mid slope compared to the foot slope and the upper slope. Soil deposition was highest on the footslope and upper slope.

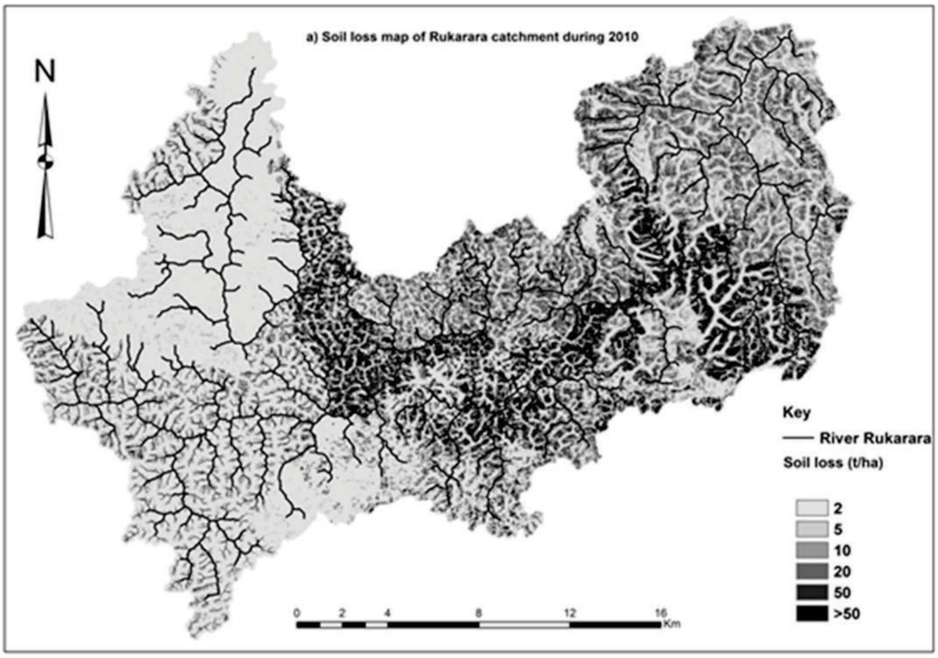

Figure 4.14a: Soil loss map of Rukarara catchment during 2010

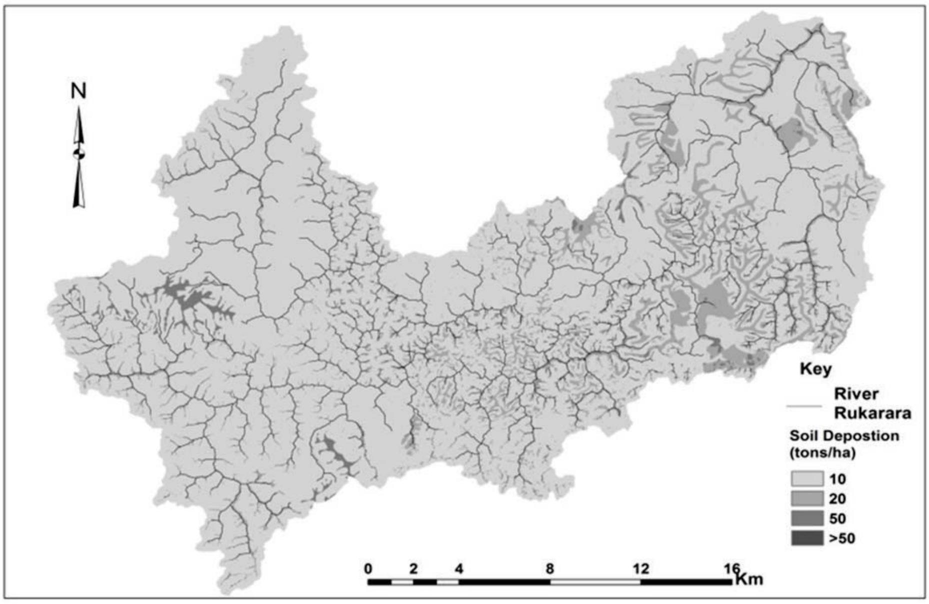

Figure 4.14b: Soil deposition map of Rukarara catchment during 2010

Table 4.8: Mean soil loss (t $ha^{-1}yr^{-1}$) by land use types and slope position in Rukarara catchment during 2010

Land use type	Footslope	Mid slope	Upper slope
Annual crops	11	25	16
Banana-coffee	13	13	13
Degraded Forest	6	4	4
Natural Forest	5	2	3
Plantation Forest	2	3	3
Savannas	6	3	4
Tea	8	8	6
Mean	5	8	7

Aggregated soil loss predicted by land use type and slope position is presented in figure 4.15. Soil loss was high in the upper slope followed by mid slope and lower slope for annual crops. The upper slope is the most cultivated. Annual crops on mid slope segments produced most soil loss expressed per hectare, but aggregated over the while catchment most sediment was generated at the hilltops and upper slope segments as most annual crops are situated in these areas. Annual crops and natural forest land use types cover large areas of the hilltops in catchment (see Figure 4.16).

4.10.2 Analysis of mean and aggregate soil loss at field and catchment scale

The results of simulated mean soil loss at field and catchment scale are presented in Table 4.9. Overall, mean soil loss was highest for annual crops, banana-coffee and tea crops, in declining order and lowest under forest and savannas land cover types. A comparison of the quantity of gross erosion at field scale and sediment load at the catchment outlet reveals that sediment load in the year 2010 was lower than gross field erosion within the catchment. About 27% of the gross field erosion is measured at the outlet. This gives a 'Sediment delivery ratio' (SDR) was 27%. SDR is defined as the total suspended solid (TSS) load at catchment outlet divided by the gross erosion within the catchment, both expressed per unit area. The difference between the level of gross erosion and TSS load leaving the catchment is a good indicator that not all the soil loss from fields reaches that catchment outlet due to sediment deposition along the route from field to catchment outlet (see figure 4.14b). The 27% value is relatively low, at least in part, the buffering capacity associated with the catchment due to the presence of plot boundary obstacles and vegetation-induced sediment deposition (Figure 4.1a and b). It is clear that the catchment possess considerable capacity to buffer sediment transport. Most of this material was stored within the catchment.

Table 4.9: Modelled mean soil loss by land use type in Rukarara catchment during 2010

Land use type	Mean soil loss (t/ha/yr)	Aggregate soil loss (tons)	% of total soil loss	Area (Ha)
Plantation Forest	3	13713	2	4839
Natural Forest	4	45456	8	12134
Degraded Forest	4	17246	3	3848
Savannas	5	1043	0	203
Tea	7	18498	3	2757
Banana-coffee	13	28214	5	2194
Annual crops	17	427061	77	24615
Field	11	551232	100	50590
Catchment scale	3	(TSS = 129518, SDR = 27%)	-	50590

SDRD = sediment delivery ratio, TSS = total suspended solids

In comparison to soil loss reported in Lake Victoria basin and particularly Rwanda (See Table 4.10), data points to a large variation in the erosion rates measured. Our study compares well with findings of Lewis, (1988) in Rwanda and results from World Agroforestry Centre, (2006) for the Nyando River Basin in Lake Victoria basin. In a nationwide study during 1983 – 84, Lewis, (1988) collected data on soil loss for 100 agricultural fields in Rwanda using soil traps and recorded a mean soil loss of 17 t ha^{-1} yr^{-1} at national scale. In the same study, mean soil loss ranged from 0 – 17 t ha^{-1} yr^{-1} for Rukarara region of south west Rwanda. Net erosion rate estimates in Lake Victoria basin part of western Kenya in the Nyando River Basin predicted for the 1963 – 2000 was 8.83 (Mg ha^{-1} yr^{-1}) spread over a range between 3.8 – 27.5 Mg ha^{-1} yr^{-1} (World Agroforestry Centre, 2006). Higher soil loss values are reported for other studies in the region (Table 4.10). For example, Kagabo et al., (2013) reported the highest mean annual soil loss of 41.5 t ha^{-1} yr^{-1} recorded with plots receiving no soil conservation practices. The high variability between simulated soil loss and measured soil loss data from field based studies in Lake Victoria basin could be due to method of data collection. Field plot data was always collected by closed erosion plots and results extrapolated to field scale. Erosion data from these studies appears to be high and exaggerated. One of the key questions in the design of experimental set-ups for erosion studies is the degree to which the plots account for issues such as connectivity and interaction between erosion patterns. Data collect by closed erosion plots lack input of transported material from outside the plot (Boix-Fayos et al., 2006). Field plots with closed boundaries affect the connectivity between preferential paths of water and sediment, and depending on the size, represent a behaviour only valid for a certain scale, not accounting for wide complexities and the interactions between different types of surface at other scales (Boix-Fayos et al. 2006, Jetten et al. 2011). When measuring erosion related processes in small field plots, especially closed plots, the connectivity of the system is broken, or at least the connectivity of the hillslope as a whole is not reflected. This is why the results of small plots of erodible surfaces (sources of runoff and sediments) are not suitable for extrapolation to smaller scales, such as hillslope or catchment without assuming some transport and deposition mechanism (Jetten, et al. 2003; Boix-Fayos et al., 2006). Connectivity aspects of landscapes are a crucial issue when linking upland area processes of sediment production with channel processes of sediment transport to catchment outlet (Helming et al. 2005).

Table 4.10: Soil losses by water erosion based on plot and modeling studies in the upstream of Lake Victoria catchment

Land cover type	Reported field erosion rate in Upstream of Lake Victoria basin (Rwanda and Burundi) (t/ha/year)		
Land use type	*Observed soil loss by field plot measurement in Rwanda (Lewis, 1988; Roose and Ndayizigiye, 1997; Kagabo, et al. 2013)*		*Predicted(new spatially explicit hydrological model in 2010)*
Cassava, sweet potatoes, maize, beans, sorghum, peas	0 - 150		17
Banana (exported residue, 10t/ha/year)	20-60		12
Pine forest (5 -10 t/ha/year of litter) or woodlots, grasslands	0.1-1		3 - 6
Land use type	*Reported field erosion rate in Downstream of Lake Victoria basin / lake periphery (Uganda)*	*References*	*Predicted (new spatially explicit hydrological in 2010)*
Annual crops	17.0 – 86.8	Bagoora (1997), Lufafa et al.,(2003), Mwanjaliwa (2004), Mulebeke (2004), Isabirye, 2005	12
Rangelands/ savannas	3.2 – 53.2	Mulebeke (2004), Mwanjaliwa (2004)	6
Coffee	19.6 – 44.9	Mwanjaliwa (2004), Mulebeke (2004)	12
Banana	25.1 – 27.9	Lufafa et al. (2003), Mulebeke (2004), Mwanjaliwa (2004)	12

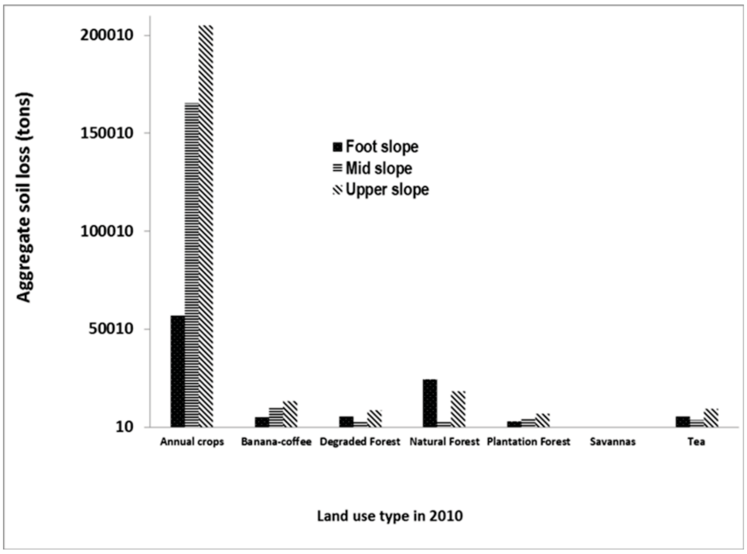

Figure 4.15: Aggregate soil loss by land use types and slope position in Rukarara catchment during 2010

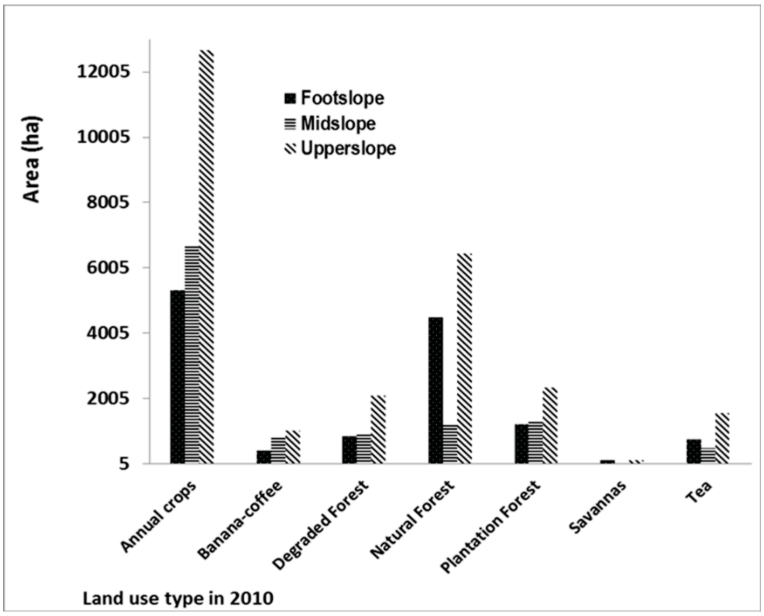

Figure 4.16: Land area of land use types at slope positions of Rukarara Catchment during 2010

4.11 Discussion

4.12 Hydrological simulation and model performance

4.12.1 Model performance

The observed and simulated run-off and baseflow show that the hydrological processes are modeled realistically. The model is calibrated to reproduce the observed hydrograph with Nash-Sutcliffe efficiency of 68% and 65% respectively for calibration and verification periods. Other publications on calibration and validation quote efficiencies ranging from 23 to 70% (e.g., Srinivasan et al. 1998; Shanti et al., 2001; Lenhart et al. 2003; Chu et al. 2004; Alansi et al. 2009; Rathjens and Oppelt, 2012). Hence, we conclude that the new model is performing rather well and can be used for hydrological and soil loss predictions. In the introduction a number of problems are mentioned from which spatial hydrological models suffer. This model is no exception.

- Apart from model limitations there are also data limitations, e.g., parameter imbalance: some data layers are detailed (e.g. DEM and land cover) and others are oversimplified (e.g. assuming homogeneous rainfall). For some of the measured events of peakflow and rainfall did not coincide, meaning that the spatial rainfall patterns that cause this peakflow was not captured by the three raingauge measurements located in the catchment. There were some cases where simulated discharge peaks seem to arrive later than might indicate. This may arise from the way rainfall data is collected, where rainfall of today is read in the next morning. In terms of input parameters the soil depth was essentially unknown and furthermore the crop growth was generalized into a single change of crop cover in the growing seasons for all annual crops. This could be improved with better field and satellite data. More complex models need more data and are more affected by this.
- This model also suffers from equifinality. However we try to limit the degrees of freedom in calibration by splitting the discharge in peakflow and basefow and coupling these directly to runoff and groundwater flow. These then determine the other fluxes in the unsaturated and saturated zone compartments. Nevertheless, the area is spatially very complex and different patterns and combination of parameters will result in the same discharge.
- The temporal resolution determines certain process descriptions: the infiltration and runoff processes are not simulated in great detail (e.g., seconds or hourly) because it uses a daily timestep. Infiltration and runoff are relatively fast processes, and generally much faster than one

day. In our model we assume that the runoff wave happens within one day which is not always correct.

- Peakflow and discharge aggregated to weekly totals are better simulated than daily values. This is confirmed by the Nash and Sutcliffe coefficients and coefficient of determination. Short time or daily periods could be simulated better with further calibration.
- Unrealistic data needs: here the model functions well. Probably soil depth is one of the most difficult parameters to obtain on this scale, while it is a sensitive parameter for the water balance, determining the soil moisture storage capacity. Most other parameters can be obtained in a reasonable amount of spatial detail. The duration of runoff activities is a second large assumption in the erosion equations which cannot be known exactly in reality.

4.13 Soil loss on different scales

Soil erosion seemed to be less severe than anticipated. This is not surprising because spatially explicit distributed erosion modeling accounts for catchment heterogeneity (sources and sinks, soil properties and agricultural practices) in sediment production, transport and deposition process and get a realistic soil loss estimates. As can be observed from Table 4.10, erosion estimates from plot studies are bound to be higher than simulated with a distributed model because plot studies generalize erosion rates for large fields but ignoring heterogeneity of these fields. Spatially, soil loss was high in the mid slope compared to the foot slope and the upper slope. This is similar to finding of Kagabo et al., (2013) in the north-western highlands of Rwanda where higher (42 t ha^{-1} yr^{-1}) and lower (16 t ha^{-1} yr^{-1}) rates of soil erosion were recorded on the Upper- slope and the Footslope, respectively. In their study, Fox and Bryan, (2000) reported that sediment peak increases with increasing slope gradient and downslope splash erosion never accounted for more than 20% of the total erosion. The sediment is usually deposited on areas of flatter slopes (e.g, Figure 4.14b). Yair and Raz-Yassif (2004) argue that there is a positive relationship between slope angle and sediment peak. Bradford and Foster, (1996) explained that when detachment is limiting, the relationship of soil loss to slope steepness is independent of soil, provided important soil properties such as erodibility or infiltration characteristics do not change with slope. This was in conclusion from their finding that change in splash with increased slope steepness was correlated *(r* = 0.96) with change in sediment yield. Hill slope position plays a major role in surface hydraulic gradients, e.g., erosion rates can increase by as much as 60 times under seepage conditions representative of midslope slope elements compared to drainage conditions that generally occur on upper slope units (Gabbard et al.1998).

Mean TSS rates for Rukarara catchment was 3 t $ha^{-1}yr^{-1}$ and was comparable with estimates from other tropical catchments. For example, estimates in

Tigray-northern Ethiopia (Haregewey et al. 2008) varied from 4 to 15 t ha^{-1} yr^{-1} with average of 9.89 t ha^{-1} yr^{-1} for 11 reservoirs (72 to 1430 ha). The tendency was for small catchment having high sediments rates and a reverse true for the big ones as there are many sinks of sediment with for instance flood plains that capture sediment.

4.14 Some issues on scaling up soil loss from field to catchment scale

In our study, it was observed that sediment load from catchment was clearly lower than gross erosion within the catchment (Table 4.9). The spatial organization of land use in the catchment may affect the connection between erosion zones and stream (Boix-Fayos et al., 2006). This catchment is characterized by mosaic vegetation and grass strips located at the lower boundary of cropped fields, making sediment delivery to the stream difficult (Figure 1a and b). This kind of field arrangement will trap some sediment in the field and reduce on the sediment reaching the catchment outlet. This buffering capacity of this catchment is closely related to the sediment delivery ratio of 27%. As observed by other researchers, high soil loss from the catchment occurs when there is a good connection between the erosion zones and the catchment outlet (e.g., de Vente et al. 2006). An earlier study at Corbeira has shown similar values of soil erosion and sediment load when the erosion area had good connection with the stream (Rodríguez-Blanco et al. 2010). This may be due to deposition occurring between the fields and stream (Walling, 1983; Lane et al., 1997; Le Bissonnais et al., 1998). Boix-Fayos et al. (2006) has explained that sediments may be deposited within the field, e.g., at the end of rills, channels and ephemeral gullies. In other cases, sediments left the fields and were deposited on adjacent fields with dense vegetation cover or in ditches and on public roads. Only some fields deliver water and sediment directly to the stream.

Another factor that could explain the differences between soil loss within the catchment and suspended sediment load at the catchment outlet are the hydrological and geomorphical processes that control the delivery, transport and storage of sediment which are highly scale-dependent (Walling, 1983; Osterkamp and Toy, 1997; Cammeraat, 2002, 2004; de Vente and Poesen; 2005). The controlling processes of sediment movement change markedly in the downstream direction as watershed area increases and the average surface slope decreases (Osterkamp and Toy, 1997). The different geomorphological processes predominating at different scales of soil erosion measurements are related across scales and different controls appear at each scale level (Boix-Fayos et al., 2006). For drainage areas between small catchments and large basins (>10 Km^2), the effect of sediment sinks often becomes dominant over sediment sources, resulting in a gradual decline in sediment yield. So, the larger the area, the greater the likelihood of sediment deposition on the way, which may even result in lower sediment yields at the basin outlet compared with the erosion rates measured on-site in erosion plots (Boix-Fayos et al. 2006). According to Cammeraat (2002) a linear upscaling from fine to broad scale is impossible, as many thresholds and non-linear processes are involved at specific scales and also at the connection between scales. In upscaling from field to catchment scale, it is important to recognize that the river floodplains which leading to the main channel systems in larger river basins can represent an increasingly important sediment sink, which will attenuate changes in upstream sediment inputs (see deposition map of Rukarara catchment Figure 4.14b).

SDR value recorded in our study (27%) is in the range of 8.43 – 39.5% reported for Nyando River Basin of the Lake Victoria basin for the period between 1963 – 2000 (World Agroforestry Centre, 2006). There are other examples of rivers where sediment yields are high or low in recent decades (e.g., by (Abernethy, 1990). Trimble (1983) found that sediment yield may be higher than erosion rates on fields if the source of sediments in the river is erosion of river banks or flood. Dedkov (2004) in Corbeira catchment reported SDR of 22 % where there are no signs of significant river bank erosion and upland erosion is responsible for catchment sediment yield. A SDR of 18 % was reported by Van Rompaey **(**2001) for 500 km^2 catchment, which is almost the size if Rukarara catchment. In the Maryland piedmont, Costa (1975) estimated that 52% of soil eroded between 1800 and 1950 was deposited as colluvium, 14% was deposited as alluvium, and 34% was exported. In Coon Creek, Wisconsin, Trimble (1983) found that, for the period 1853–1977, 49% of sediment was deposited as colluvium, 45% in alluvial storages, and 6% was exported. A high SDR value may mean a low degree of buffering found in catchment, little sediment will be deposited and there will be only limited sediment available to be remobilized during times of reduced supply. The reasons for high SDR could be due to high drainage density and the frequent occurrence of hyper concentrated flows caused by big intensive storms, which reduce the potential for sediment deposition. The occurrences of steeps slopes that are intensively cultivated and poor highly erodible soils could be another cause of high SDR. Regions which are heavily impacted by land use activities and soil erosion, the sediment delivery ratios ranged from zero to 89% (Walling, 1999). Such river basins can therefore be expected to be poorly buffered and the sediment output will be sensitive to changing erosion rates associated with changing land use (Walling, 1999).

4.15 Conclusion

The new spatially explicit hydrological model is able to reproduce the observed hydrograph with Nash-Sutcliffe efficiency of 68% and 65% respectively for calibration and verification periods. The predictive capability of reproducing field situation of peakflow, discharge and sediment delivery by surface runoff from hillslopes to river is demonstrated as being reasonable. This model does not simulate the infiltration and runoff process in great detail (e.g., seconds or hourly) because it uses a daily timestep. Infiltration and runoff are relatively fast processes, and generally much faster than one day. In our model we assume that the runoff wave happens within one day. Weekly peak flow and discharge are better simulated than results of daily periods. Short time or daily periods could be simulated better with further calibration. Despite problems related with the spatially distributed parameters (rainfall and soil depth) for calibration of the model associated with to equifinality, the good model performance suggests that we found an adequate parameter set for calibration. The model is capable of producing field level sediment dynamics which are in line with soil loss found in field studies in similar environments in Rwanda and the Lake Victoria basin. Using variables such as rill density and duration of runoff the overall erosion intensity of the model can be calibrated, which resulted in acceptable and logical soil loss patterns. However we also realize that in reality, the sediment dynamics are in part determined by the actual connectivity between fields. Many details such as footpaths, hedgerows, directions of tillage etc. that influence local soil loss could not be incorporated in the model.

It is clear that soil loss at field level cannot be extrapolated to catchment level without using either sediment delivery ratios or transport capacity based modelling. The advantage of the latter is that realistic patterns of erosion and deposition can be derived and we can couple nutrient fluxes with the sediment balance.

Annual crops on mid slope segments produced most soil loss expressed per hectare, but aggregated over the while catchment most sediment was generated at the hilltops and upper slope segments as most annual crops are situated in these areas. Banana crops were the second highest in spite of the fact that they are tree crops, because the ground cover is generally not very high in the area. Natural vegetation and plantations had the lowest soil loss values. Slope is a driving force in soil loss as can be seen from the spatial patterns, but influences mainly differences in soil loss of annual crops.

We conclude that the model is performing rather well and can be used for hydrological and soil loss predictions. Immediate improvements can be done in terms of:

- a better representation of crop growth, either by crop growth modeling, better field and satellite data; for instance NDVI time series. Crop growth was generalized into a single change of crop cover in the growing seasons for all annual crops. More complex models need more data and are more affected by this.

- the temporal resolution determines certain process descriptions: the timestep of one day prohibits the simulation of the infiltration and run-off process in detail (e.g., seconds or hourly). Infiltration and runoff are relatively fast processes, and generally much faster than one day. In our model we assume that the runoff wave happens within one day which is not always correct. Different concepts are needed. Infiltration and run-off process description could be improved and modeling with high spatial and temporal data. High temporal measurement of stream water level (e.g., seconds or hourly) using automatic pressure sensors could provide accurate data for modeling infiltration and run-off process in detail. The valley floor with groundwater tends to fill up and then remains "full", which results in little groundwater movement because there are no pressure differences. It is not known in how far this reflects the reality, although the stagnating groundwater has sustained rice production throughout the year in the flat valley bottom. Coupling between the groundwater and baseflow has to be improved but this needs a better process description and also a more detailed hydrological dataset.

4.16 Acknowledgements

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Chapter 5

Soil fertility and nutrient balances of low input land use systems of South-West Rwanda, Upstream of Lake Victoria Basin

Abstract

This study reports on a nutrient balance assessment in the south western highlands of Rwanda, a densely populated agricultural area. N, P and K stocks and flows were assessed for five different land use systems (including tea, banana, maize, beans), at both plot and farm level. The wealth level of farm families and the slope position (upper slope, mid-slope, foot slope) were used to further stratify the area. Nutrient inputs calculated or estimated include mineral and organic fertilizers, atmospheric deposition, biological nitrogen fixation and sedimentation. Nutrient outputs include removal of harvested crops and residues, leaching, gaseous losses and erosion. Sedimentation and erosion was estimated using a new approach with spatially explicit hydrological model for the feedback between erosion and sedimentation in a daily continuous way and in spatially explicit quantitative terms (at 30 m x 30 m pixel level). Overall mean farm partial balances was marginally positive (8 kg N ha^{-1} yr^{-1}, 3 kg P ha^{-1} yr^{-1} and 8 kg K ha^{-1} yr^{-1}) and was low among food crops. N and K overall mean full balances were generally negative across land use systems while P was positive (– 36 kg N $ha^{-1}yr^{-1}$, +2 kg P $ha^{-1}yr^{-1}$, -8 kg K $ha^{-1}yr^{-1}$). Among wealth groups, nutrient losses were highest for rich farmers (-42 kg N $ha^{-1}yr^{-1}$, -0.1 P $ha^{-1}yr^{-1}$, -20 K $ha^{-1}yr^{-1}$) compared to medium (-33 kg N $ha^{-1}yr^{-1}$, +3.5 kg P $ha^{-1}yr^{-1}$, -11 kg K $ha^{-1}yr^{-1}$) and poor farmers (-30 kg N $ha^{-1}yr^{-1}$, +1.1 kg P $ha^{-1}yr^{-1}$, -8 kg K $ha^{-1}yr^{-1}$). More nutrient losses occurred on mid-slope positions (-37 kg N $ha^{-1}yr^{-1}$, +3 kg P $ha^{-1}yr^{-1}$, -11 kg K $ha^{-1}yr^{-1}$) compared to the upper slope (-35 kg N $ha^{-1}yr^{-1}$, +1 kg P $ha^{-1}yr^{-1}$, -17 kg K $ha^{-1}yr^{-1}$) or foot slope (-30 kg N $ha^{-1}yr^{-1}$, +2 kg P $ha^{-1}yr^{-1}$, -3 kg K $ha^{-1}yr^{-1}$). The trend of N mining was highest for napier grass > tea > banana-cereal > fallow > cereals > root – tuber > banana – coffee > banana – bean > legume – banana, in declining order, respectively. A positive P and K balance was only observed in the intensively managed banana – coffee and tea crops land use systems. Erosion contributed the highest loss for P (85%) (7 kg $ha^{-1}yr^{-1}$) and 88% K (72 kg $ha^{-1}yr^{-1}$) in the nutrient balance. Generally, there is nutrient mining in south west Rwanda agricultural systems since soil nutrient stock has to offset the negative balances each year. In relation to the nutrient stocks the results are less alarming for all nutrients with depletion rates of about 1% per year. This study highlights the need to consider soil fertility restoration through integrated soil fertility management to improve nutrient capital on smallholder farms for sustainable food production in south west Rwanda.

***Keywords**: Land degradation; Nutrient balances; Soil fertility decline; Soil erosion; Food security; Sustainable agriculture; Rwanda*

5.1 Introduction

Food insecurity is a major problem in many Sub Saharan African countries. Levels of crop productivity are low for much of sub-Saharan Africa, and nutrient balances are typically negative (Wortmann and Kaizzi 1998; Smaling et al. 2012). Rwanda imports 36 % of its food requirements (Diao et al. 2012). About 51% of all households reported some type of difficulty accessing food, 14% of households have usual and almost year round chronic difficulties in accessing food and a total of 21% of households are food insecure (Vinck et al. 2009). Rwanda's population is increasing at rate of 2.9% per annum, food demand is rising and will continue to rise as a result. Crop yields are low, stagnated over the years (FAOSTAT 2008) and food production has declined per capita (Olson 1994; Ansoms 2009; Nabahungu 2012). For example, the potential yields of 5 t ha^{-1} for maize grain and 23 t ha^{-1} for potato, the actual yields are 1.5 t ha^{-1} and 14 t ha^{-1} for maize grain and potato, respectively (NISR, 2010). Soil fertility decline may be one of the causes of increased dependence on foreign food imports but accurate data to support this claim is missing. Following its continental assessment (Stoorvogel et al. 1993), several studies have revealed that soil fertility decline is one of the principal causes for low agricultural productivity and food insecurity in Africa (e.g., Smaling, 1993; Nandwa and Bekunda, 1998; Sanchez, 2002). This is illustrated by small increases in crop productivity, even in years with adequate rainfall (Haileslassie et al. 2005). In spite of the limitations that are inherent to its country and continental scale assessment of soil fertility decline in Africa (Stoorvogel et al. 1993), this study brought to prominence the large and continuous soil nutrient depletion facing many smallholder African farmers to date and its likely constraint on future food production. Their study estimated nutrient balances for Sub Saharan Africa (SSA) for the period 1982-84 and projections for 2000 are -22 and -26 kg N ha^{-1} yr^{-1}; -2.5 and -3.0 kg P ha^{-1} yr^{-1}; and -15 and -19 kg K ha^{-1} yr^{-1}, respectively. The study triggered numerous case studies at plot, farm, village, district and national levels. A recent review by Cobo et al. (2010) provided a summary of 57 nutrient balance case studies in Africa. This review suggests that it has been in Kenya where most of the research on nutrient balances has been carried out (19 out of 57 studies), which is more than two times than in the succeeding countries, Ethiopia, Mali and Uganda. The authors reported that most farming systems had negative N and K balances (i.e. 85 % and 76%, respectively). For P the trend was less noteworthy (i.e. only 56% of studies presented means below zero). Intensively cultivated highlands in East Africa lose an estimated 36 kg N ha^{-1} yr^{-1}, 5 kg P ha^{-1} yr^{-1}, and 25 kg K ha^{-1} yr^{-1}. In this review, only few studies were available for the Lake Victoria basin (e.g., Esilaba et al. 2005; Wortmann and Kaizzi 1998).

Many studies report land degradation in Rwanda, especially in the highland areas in the upper catchment of Lake Victoria basin (e.g., Olson 1994;

Verdoodt and Van Ranst 2006; Roose and Ndayizigiye, 2007; Kagera Basin Monograph 2008; Ansoms et al. 2008, and Ansoms 2009; Musahara and Rao 2009). Nutrient balance studies as a tool to assess the sustainability of agro-ecosystems and potential consequences for agricultural productivity are scanty in Rwanda. As part of the African study at a coarse scale, Stoorvogel et al. (1993) reported large national negative nutrient balances for Rwanda for the period 1982- 84 and projections for 2000 are -54 and -60 kg N ha^{-1} yr^{-1}; -9 and -11 kg P ha^{-1} yr^{-1}; and -47 and -61 kg K ha^{-1} yr^{-1}. There is uncertainty related to these figures given the coarse scale on which these values were modeled. Since there is large variability in soil fertility and also differences in nutrient management between and within farms in Rwanda (Kagabo et al. 2013; Roose and Barthès 2001), nutrient balance may be closely related to farm type, crop type, farm wealth category and slope position depending on the way each particular field has been managed over decades. This results in a mosaic of degrees of nutrient mining at the landscape scale (Smaling 1993; Van den Bosch et al. 1998). Therefore, improved understanding of biophysical variability and functioning of smallholder farming systems is required for improved targeting of soil fertility management interventions (Zingore 2006). A study by Nabahungu (2012) in Rwanda showed that partial N, P and K balances were negative for crop plots for the wetland and hillside fields of the Cyabayaga catchment (eastern savannah agro-ecological zone in the Eastern province) and the Rugeramigozi wetland area (plateau agro-ecological zone of the Southern province) of Rwanda. There are no nutrient balance studies for the south west highlands of Rwanda. Wasige et al (2013a) only documents land use changes in the upper catchment of Lake Victoria basin, Rwanda, but didn't investigate the effects of these changes on nutrient balances. They found that the most dominant land use change processes were gains in farmland areas (60% by 2010) and a net reduction in dense forest (7 % to 2.6 %). Another study by Wasige et al. (2013b) in south west Rwanda, focused only on soil organic carbon (SOC) that was found to be towards the low end of the spectrum (2.5 % on average) for agricultural land uses compared to about 12 % in the natural forest land use. The declining SOC stocks in the agricultural land uses indicate low soil fertility status that needs to be evaluated since SOC is a proxy for soil fertility (Smaling and Dixon 2006). The objective of this study was to assess how current farming affects nutrient stocks in south west Rwanda, i.e., do they further decrease (negative net flows) or not? And if so, which flows are dominant? We hypothesized that traditional agricultural production systems and technologies in south west Rwanda promote soil fertility mining on smallholder farms. N, P and K stocks and flows were assessed for five different land use systems, at both plot and farm level. The wealth level of farm families and the slope position (upslope, mid slope, and foot slope) were used to further stratify the area. Nutrient inputs calculated include mineral and organic fertilizers, atmospheric deposition, biological nitrogen fixation and sedimentation. Nutrient outputs include removal of harvested crops and residues, leaching, gaseous losses and erosion. A new spatially explicit hydrological model by Wasige et al. (2013c) was applied to quantify the feedback between erosion and sedimentation at pixel level (30 m x 30 m) to provide results at high spatial detail. The resolution of 30x30 m is in the order of agricultural field size (0.01 – 1.2 ha) in south west Rwanda. This model requires relatively little data and has a better capability of predicting and evaluating the effects land use change on soil erosion from field to catchment scales compared to the current soil erosion models used in nutrient balance studies e.g., the USLE (universal soil loss equation) by Stoorvogel et al. (1993) and LAPSUS (LandscApe ProcessS modeling at mUltidimensions and Scales) by Lesschen et al. (2005).

5.2 Materials and methods

5.2.1 Site selection and description of study area

The study site is the 506 km^2 river Rukarara catchment in the hill slopes of south west Rwanda and in the upper reaches of the Nile basin, also known as the "water tower" of lake Victoria and a source for the river nile (2^{0}28'-2^{0}34' S and 29^0 23'-29^{0}29'E). The river has its source at about 3000 metres above sea level (m.a.s.l) and descends to 1470 m at the outlet. The study site was selected being an area where the steep slopes have lost most of their forest cover, a densely populated agricultural area under intensive and long-term cultivation (Wasige et al. 2013a) and are experiencing high levels of soil erosion (Kagera Basin Monograph 2008, Musahara and Rao 2009). The low land is hotter and drier than the upstream area with temperature ranging from 15^{0}C - 25^{0}C. The rain pattern is bi-modal, with long rains occurring during September to January, and shorter rains in March to June. Annual precipitation ranges from 1200 to over 1800 mm per annum. Hill-slopes exhibit physical variability in terms of elevation, geology, soils, climate and crop production. The geology of the area consists of bedrock formed in the middle Precambrian period. The catchment is fully forested on it's upper part (Nyungwe natural forest) while the middle to lower parts of the catchment is mainly covered by agricultural crops. As rainfall is bimodal, farmers grow two annual crops per year. Hillslopes are overlayed uniquely by five land use systems from upstream to downstream (see Table 5.1 and Figure 5.1) as follows; (1) Tea-Annual and (2) Mixed Banana-Annual Land use systems with slight higher inputs applied in soil fertility management occur in a high rainfall zone. The other three land use systems include; (3) Mixed Banana-Coffee-Annual land use system, (4) Cassava-Sorghum-Bean land use system (5) Banana-Cassava-Sorghum land use system occur under low input and low rainfall. Inter-cropping systems are widely practiced combining different crop mixes (between 7 and 8 on crops on average) on all slope positions and usually intercropped with beans. Major crops include; beans; bananas; sweet potatoes; maize; cassava, sorghum and tea at the subsistence level of less than 1 ha with nearly no or limited fallow. Tea, banana, coffee and maize are the major commercial crops. Tillage is mainly undertaken with the use of simple hand implements. Overcultivation with limited soil fertility inputs is one major concern for soil degradation in respect to soil nutrient depletion. Some farmers practice terracing but there is lack of maintenance. The land use systems integrate both crops and limited livestock component.

5.2.2 Data collection and quantification of Nutrient Budgets

5.2.2.1 Household selection and field sampling points identification

Based on classification scheme of farm wealth classes developed by Nabahungu and Visser (2011) for smallholder crop production systems in different parts of Rwanda and with help of community leaders, stratified toposequential randomized sampling approach was used to select household in each land use system across three farm family wealth classes and slope positions. The wealth classes were categorized as follows: poor (less than 0.25 ha, no cow); medium (>0.25<0.5 ha, 1 cow), rich (> 0.5 ha, >1 cow). Slope positions were categorized as follows (see Figure 5.2): footslope (0–8% slope), steep mid slope (>15% slope) and upper slope (8–15% slope) (Wang et al. 2001; Kagabo et al. 2013). Slope position was derived from an ASTER DEM (30 m x 30 m) using the Topographic Position Index (TPI) tool in ArcGIS10. The output map was overlaid on a village map and sampling locations were identified in ArcGIS10. A field sampling map was then printed to support field identification of selected locations of households in the land use system. Given the logistical limitations and required monitoring frequency, a total of 91 farm households spread over five land use systems were randomly selected to participate in the study. To identify the various field plots, land use map was constructed by the farmers with help of trained graduate research assistants and the fields on the map were serially numbered for reference purposes during the questionnaire interview. Farm fields were mapped (Figure 5.1) with a Geographic Positioning System (GPS). Farmers owned several plots ranging from 3 to 16 per farm. Individual household plots were grouped into a farm depending on the existing land use system (coded as T, R1, R2, R3 and R4; see Table 5.1). Identifying, quantifying and recording of the resource flows into and out of the farms and distribution within the farms was done by farmers under supervision of a research assistant. These resource flows data were captured during a two-time survey at the end of each of the two land use seasons of 2010/2011 using semi-structured questionnaire interviews covering major aspects of land, crop and livestock management based on Nutrient Monitoring (NUTMON) tool box (Vlaming et al. 2001).

Table 5.1. Land use systems and farm typologies characterization*

Characteristics	T	R1	R2	R3	R4
Precipitation (mm yr^{-1})	1608	1608	1332	1170	1310
Potential Evapotranspiration (mm yr^{-1})	831	831	927	1031	1560
Temperature (^{0}C)	13-23	13-23	12-24	13-25	13-29
Elevation (m)	2100	2000	1700	1600	1500
Major soil units (FAO, 1988)	Cambisols, Ferralsols, Gleysols Histosols	Alisols	Acrisols Cambisols,	Cambisols, Lixisols	Lixisols Cambisols
Proportion of crops in the land use system	40 % Tea and 60 % Annual crops	30 % Banana and 70 % Annual crops	30 % Banana, 20 % Coffee and 50 % Annual crop	30 % Cassava, 40 % Sorghum and 30 % Bean	30 % Banana, 30 % Cassava and 40 % Sorghum
Farm production orientation	cash crop	Home consumption and cash crop	Coffee (cash crop), banana-cereals for home consumption and cash crop	Home consumption and cash crop	Home consumption and cash crop

* T. Tea-Annual, R1. Banana-Annual, R2. Banana-Coffee-Annual, R3. Cassava-Sorghum-Bean, R4. Banana-Cassava-Sorghum

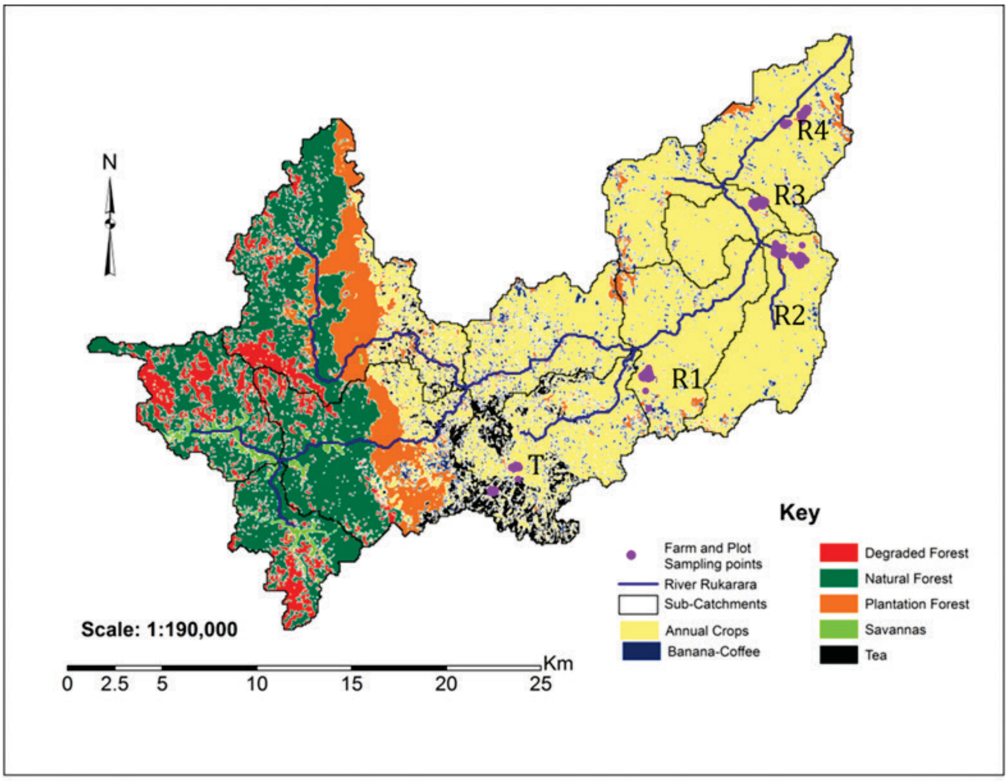

Figure 5.1: Map showing Location of sampled farms in Rukarara catchment (T. Tea-Annual, R1. Banana-Annual, R2. Banana-Coffee-Annual, R3. Cassava-Sorghum-Bean and R4. Banana-Cassava-Sorghum)

Soil, manure and crop sampling

Data collected included: soil, organic inputs and crop samples from farm households (Figure 5.1) to obtain knowledge of the current level of soil fertility and also to provide background information on nutrients stocks. A total of 506 soil composite samples were taken from 0 to 20 cm soil depth of farm plots and major local soil units. Additional 21 soil samples were collected from adjacent forest land use systems for comparison with the current level of soil fertility in land use systems. Soil analyses were carried out following standard procedures for tropical soils as described in Anderson and Ingram (1993) and Okalebo et al. (2002). Other data types collected included; field area sizes, undisturbed soil cores for determination of soil bulk densities, composite samples of five sub-samples from manure and crop products were collected. Dry matter yield and total N, P and K contents were determined by laboratory analyses (Table 5.2). Secondary data were collected were mainly from related publications on smallholder farm nutrient balances in the highlands of East Africa (see Table 5.2). Rainfall data for 2010/2011 land use period was provided by the Rwandan meteorological services. The data collected was used to quantify nutrient flows and balances by multiplying the mean mass dry matter fractions of the resources with their nutrient contents. Soil nutrient stocks were calculated from soil nutrient concentrations and bulk densities. Nutrient depletion rate was calculated as a percentage of net nutrient balance to the current nutrient stocks.

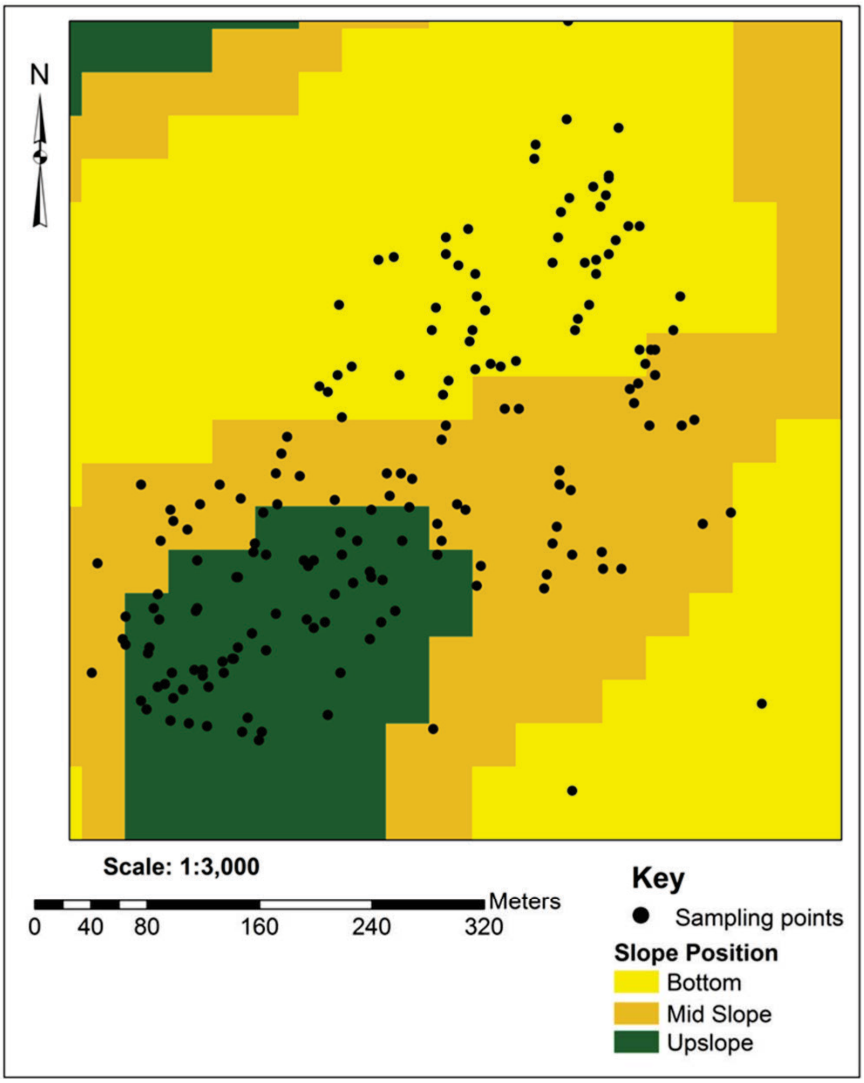

Figure 5.2: Map showing the location of plot sampling points according to slope position

Table 5.2: Nutrient fractions (%) of crop products, crop residues, manure, external inputs that were used in the construction of nutrient balances

Resource	Nutrient concentration (%)						Reference
	Commodities/grain/fruit			Crop residue/ waste			
	N	P	K	N	P	K	
Farm yard manure (FYM)	-	-	-	0.41	0.10	0.5	Laboratory analysis
Compost	-	-	-	0.61	0.10	0.50	Laboratory analysis
Household waste	-	-	-	1.2	0.2	1.0	Laboratory analysis
Cattle manure	-	-	-	1.7	0.8	1.3	Laboratory analysis
Goats manure	-	-	-	1.4	0.4	2.2	Laboratory analysis
Poultry manure	-	-	-	4.5	0.5	1.2	Laboratory analysis
Sweet Potatoes	3.6	0.29	2.4	-	-	-	Laboratory analysis
Tea	3.8	0.26	2.3	-	-	-	Laboratory analysis
NPK fertilizer	17	17	17	-	-	-	Factory value
DAP fertilizer	18	20	-	-	-	-	Factory value
Urea fertilizer	46	0	0	-	-	-	Factory value
Wheat grains	2.0	0.39	0.44	-	-	-	Kent (1983)
Banana fruit	0.26	0.01	0.90	1.0	0.15	0.20	Wortmann and Kaizzi (1998)
Sweet Potatoes	-	-	-	3.1	0.16	2.4	Van de Bosch et al. (1998)
Bean	4.1	0.32	2.6	4.4	0.36	2.8	Van de Bosch et al. (1998)
Coffee	2.5	0.15	2.1	2. 9	0.17	1.8	Van de Bosch et al. (1998)
Maize grains	2.3	0. 22	1.9	2.1	0.2	2.1	Van de Bosch et al. (1998)
Cassava	3.3	0.19	2.1	4.5	0.27	2.6	Van de Bosch et al. (1998)
Green grams	4.0	0.38	1.1	4.4	0.04	0.5	Van de Bosch et al. (1998)
Pigeon peas	4.0	0.29	1.8	4.1	0.22	1.5	Van de Bosch et al. (1998)
Millet grains	2.3	0.25	1.6	1.3	0.15	1.8	Van de Bosch et al. (1998)
Napier grass	-	-	-	2.3	0.24	3.0	Van de Bosch et al. (1998)
Irish potatoes	0.34	0.23	3.1	1.4	0.33	0.0	Van de Bosch et al. (1998)
Sorghum	1.7	0.23	0.73	2.1	0.21	1.6	Van de Bosch et al. (1998)

Table 5.3: Data sets and model improvement used in the nutrient balance calculation in south west Rwanda

Land use system definition	The dominant land use types were identified in the study site during field visits (see Table 1)
Nutrient stocks	Calculated from laboratory analyzed soils samples
IN1: mineral fertilizer	Data obtained per crop and total consumption from a household survey conducted during the cropping period
IN2: organic inputs	Data obtained from a household survey conducted during the cropping period, in combination with literature data on nutrient contents
IN3: atmospheric deposition	Defined by a regression model function linking nutrient input with mean annual rainfall (Smaling, 1993)
IN4: Biological nitrogen fixation	Defined by a regression model function linking nutrient input with mean annual rainfall (Lesschen et al. 2007)
IN5: sedimentation	Calculated using a new spatially explicit hydrological model (Wasige et al. 2013c)
OUT1: harvest products	Data obtained from a household survey conducted during the cropping period
OUT2: Other organic outputs	Data obtained from a household survey conducted during the cropping period
OUT3: leaching	Regression model based on review by De Willigen (2000)
OUT4: gaseous losses	Regression model based on data from IFA/FAO (2001)
OUT5: erosion	Erosion calculations using a new spatially explicit hydrological model (Wasige et al. 2013c)

Soil nutrient balance calculation

Full and partial nutrient balances and flows were calculated at levels of individual land use systems and farms NUTMON model v3.6. The wealth classes of farm families and the slope position were used to further stratify of the dataset. Full nutrient (FN) balances refer to the sum of all inputs minus the sum of all outputs (equation 8) from all measured and estimated fluxes. Partial nutrient (PN) balances (equation 9) only cover the easy-to-quantify flows IN 1 and 2, and OUT 1 and 2. They are clearly incomplete from a biophysical stand point, but more accurate than a full balance, and offering a basis for strategic farm decisions. The partial balance is based on the flows that are most easily managed, whereas the full balance is an indicator for soil nutrient depletion (Smaling et al. 2012). NUTMON model is parameterized according to Nutrient Monitoring (NUTMON) methodology of five major input (IN) and output fluxes (OUT) (summarized in Table 5.3) described in Smaling et al. (1993); Smaling and Fresco (1993) and more recently by Lesschen et al. (2007). The details of NUTMON equation are given below as follows;

IN1: mineral fertilizer

Nutrient input from mineral fertilizer (IN1) was available from household survey data for the management period covered in this study. Urea, NPK and DAP (diammonium phosphate) are the fertilizers commonly used.

IN2: organic inputs

The total amount of nutrients from organic fertilizers (e.g., manure, animal beddings, household waste and crop residues) was calculated by multiplying dry weight with nutrient content (see Table 5.2). Based on the livestock numbers the total amount of nutrients input from manure was calculated by multiplying the livestock densities by the excretion per animal per year and the nutrient content of the manure for each livestock class (Lesschen et al. 2007).

IN3: Atmospheric deposition

Nutrient inflows from wet atmospheric deposition (IN 3) were calculated using the transfer function, linking nutrient input with mean annual rainfall (Smaling, 1993);

$$\text{IN 3N} = 0.14p^{1/2} \tag{5.1}$$

$$\text{IN 3P} = 0.023p^{1/2} \tag{5.2}$$

$$\text{IN 3K} = 0.092p^{1/2} \tag{5.3}$$

Where, IN 3N, IN 3P, IN 3K , is the input of N, P and K (kg ha^{-1} yr^{-1}) and p is the mean annual precipitation (mm yr^{-1}).

IN 4: Biological nitrogen fixation

N fixation was calculated using the following equation (Lesschen et al. 2007);

$$N_{\text{fixed}} = 0.5 + 0.1\sqrt{rainfall} \tag{5.4}$$

with rainfall in mm $year^{-1}$ and N_{fixed} in kg N ha^{-1} $year^{-1}$.

IN5: sedimentation

Sedimentation was calculated using a new spatially explicit distributed hydrological model (Wasige et al 2013c), which simulates feedback between erosion and sedimentation at pixel level (30 m x 30 m), see details under OUT5.

OUT1: Harvest or sold products

Harvest or sold products were calculated by multiplying the dry matter yield with the nutrient content of the products (see Table 5.2).

OUT2: crop residues

The output by nutrients in crop residues exported was calculated by multiplying yield, nutrient content of the crop residues (see Table 5.2).

OUT3: leaching

Determination of nitrogen leaching was based on De Willigen's (2000) regression model. This model is based on an extensive literature search and is valid for a wide range of soils and climates (Lesschen et al. 2007);

$$\text{OUT3}_N = (0.0463+0.0037\times(R/(C\times L))\times((\text{IN1}+\text{IN2})+D\times\text{NOM-U}) \tag{5.5}$$

Where; R is rainfall (mm), C is clay content in the topsoil (%), L is the layer thickness or rooting depth (m), D is the decomposition rate (set at 1.6 % per year), NOM is the amount of N in soil organic matter (kg N ha^{-1}) and U total nutrient uptake by the crop. For potassium leaching, a new regression model developed by Lesschen et al. (2007), based on the same data set of De Willigen (2000) was used;

$$\mathrm{OUT3}_K(\mathrm{kg\ N\ ha^{-1}}) = -6.87 + 0.0117\ x\ P + 0.173\ x\ F_K - 0.265\ x\ CEC \qquad (5.6)$$

Where; P is the mean annual precipitation, F_K= mineral and organic fertilizer potassium (kg K ha^{-1} yr^{-1}) CEC is cation exchange capacity (me/100g).

OUT4: gaseous losses

We applied a regression model developed by Lesschen et al. (2007) to estimate gaseous losses for N_20, NOx and NH_3. The regression is based on literature data for tropical environments derived from a larger data set (IFA/FAO 2001). Regressions were combined into the following equation, which had an R^2 of 0.70;

$$\mathrm{OUT4\ (kg\ N\ ha^{-1}year^{-1})=0.025+0.000855xP+0.130XF+0.117xSOC} \qquad (5.7)$$

Where;
P = precipitation (mm year^{-1}), F = mineral and organic fertilizer nitrogen (kg N ha^{-1} year^{-1}), SOC = soil organic carbon content (%).

OUT5: erosion

Erosion and sedimentation was calculated using a new spatially explicit hydrological model by Wasige at al. (2013c) which simulates feedback between erosion and sedimentation at pixel level (30 m x 30 m) is used. The resolution of 30x30 m is in the order of agricultural field size (0.01 – 1.2 ha) in south west Rwanda which allowed us to couple the model to the farm nutrient balance research. The model is grid based with continuous daily water balance and good representation of sediment delivery principles to takes care of a detailed spatial variability (sources and sinks) when estimates of soil loss are made daily, throughout the growing season or annual basis. The model was calibrated to local hydrological measurements of the catchment (e.g., peak flow and discharge) measured during two year period of 2010/2011 (Wasige et al. 2013c). Simulated soil loss was calibrated using data from long terms field studies in the highlands of Rwanda (e.g., Lewis, 1988; Kagabo et al. 2013). The loss (OUT5) or gain (IN5) of nutrients was calculated by multiplying soil loss or sedimentation by the soil nutrient contents and an enrichment factor. Based on literature the enrichment factor was set to 2.3 for N, 2.8 for P and 3.2 for K (Smaling et al. 2012).

Net nutrient balance calculation

From the above calculations of five inflows and five outflows, the net nutrient balance is calculated for Nitrogen (N), phosphorun (P) and Potassium (K) macro nutrients as the total of nutrient inputs minus nutrient outputs expressed in kilogram nutrients ha^{-1} yr^{-1}. Both full and partial nutrient balances for the macronutrients of 0 – 20 cm rootable soil layer were derived. Full nutrient (FN) balances refer to the sum of all inputs minus the sum of all outputs (equation 8) from all measured and estimated fluxes. Partial nutrient (PN) balances (equation 9) only cover the easy-to-quantify flows IN 1 and 2, and OUT 1 and 2. They are clearly incomplete from a biophysical stand point, but more accurate than a full balance, and offering a basis for strategic farm decisions. The partial balance is based on the flows that are most easily managed, whereas the full balance is an indicator for soil nutrient depletion (Smaling et al. 2012).

$$FN = (IN1+IN2+IN3+IN4+IN5)-(OUT1+OUT2+OUT3+OUT4+OUT5) \quad (5.8)$$

$$PN = IN1 + IN2 - OUT1 + OUT2 \quad (5.9)$$

5.2.2.2 Statistical data analysis

The calculation of household characteristics, the first four inflows and outflows of the nutrient balance were implemented in NUTMON model version v3.6. All statistical analyses of model outputs and soil analytical data were then performed with SPSS 20 for Windows. Significant differences were tested one way analysis of variances (ANOVA) with Games-Howell Post Hoc multiple pair comparisons test and descriptive statistics at a probability level of $p < 0.05$. Data analysis detected and separated means of significantly different parameters for plots and farms across land use systems, slope position and farm family wealth class. The Games Howell (Games and Howell, 1976) test was selected for the Post Hoc comparison because it is robust for unequal sample sizes (not all combinations of farm type, crop type, wealth classes are there) with different variances across the groups, which was the case with our data set.

5.3 Results

5.3.1 Soil chemical fertility

The baseline data on soil properties of south west Rwanda under agricultural soils and forest soils is presented in Table 5.4. Soil properties indicate that these soils are acidic. Soils dominated by forest land use types have significantly richer nutrient stocks compared to agricultural land use system. SOC stocks in the top 20 cm soil depth lies between 23 to 40 gkg^{-1} i.e., nearly 30 - 50 % of the amount found in adjacent forest land use systems

(66 to 72 g kg^{-1}). Differences of similar magnitude were observed for stocks of soil nitrogen. Total soil nitrogen in agricultural land use types lies between 1.9 to 3.1 gkg^{-1} which is nearly 30% stocks found in adjacent forest land use types (9 gkg^{-1}). Total nitrogen (TN) and Olsen P were significantly different ($P<0.05$) among forest types and were highest in plantation forest. Exchangeable K was significantly higher in agricultural soils compared to forest soils. Land use systems are characterized by below critical values for soil fertility in SOC, N, K and while levels of P were above the critical level (Table 5.4). Soil nutrient concentrations were slightly higher above the critical levels in tea land use system that food land use system but was comparable with the level in forest land use systems. SOC and N were also highest in tea land use system compared to food crop land use system.

Table 5.4. Mean and standard deviations (in brackets) of nutrient concentrations (0-20 cm) of land use system and forest type

Description	T	R1	R2	R3	R4	Overall agricultural land use system mean nutrient concentrations	Natural forest	Degraded natural forest	Plantation forest
No. Samples	105	112	138	101	50	N.A	11	5	5
pH	4.4(0.6)abc	4.6(0.4)bcde	5.1(0.7)e	4.9(0.5)de	4.8(0.3)cde	4.4	3.8 (0.4)a	4.2 (0.1)ab	4.5 (0.2)bcd
SOC ($g kg^{-1}$)	40(11.4)a	31(6.7)b	23(5.4)b	26(5.8)b	32(8.6)b	30.4	72 (7)c	66 (8)c	72 (6)c
TN ($g kg^{-1}$)	3.1(0.7)a	2.5(0.5)ab	1.9(0.4)b	2.3(0.4)b	2.6(0.6)ab	2.48	9 (2)c	8 (2)d	9 (2)e
Olsen P ($mg kg^{-1}$)	37(15)a	48(26)a	54(30)a	33(11)a	35(14)a	41.4	783 (156)b	756 (232)b	869 (96)c
Exch. K ($mg kg^{-1}$)	26(21)ab	29(26)ab	35(28)ab	33(29)ab	32(39)ab	31	14 (3)c	20 (5)bc	8 (1)c

SOC = soil organic carbon; TN = total nitrogen, ^{a,b,c,d} show significant differences between land use systems ($P<0.05$). Values with similar letters are not significantly different, N.A- not applicable.

5.3.2 Farm resource holdings: Land, livestock, labour force and fertilizer resources

On average, total area of farm land per household was 0.6 ha (Table 5.5). There were significant differences in farm land area between farm family wealth classes and not by farm groups ($P<0.05$). The mean number of persons per farm household was 6. Medium and rich wealth farms had slightly more household members. Calculated man-equivalent labour force per farm was about 3 persons on average. Tea land use system had the highest labour force. Mean livestock per household was 0.2 TLU (Tropical Livestock Unit). Livestock comprises mainly zero-grazing dairy cattle and kept under zero grazing units. There were significant differences between farms with wealthy farms having higher TLU. Organic and inorganic inputs consisted of application of inorganic fertilizers, farm yard manure (FYM) and household waste to fields (Table 5.5). NPK, Urea and DAP were the only mineral fertilizers applied. Wealthy farms applied significantly more inorganic inputs than poor farm groups. On average, 7 kg of inorganic fertilizers was applied on non-tea farm groups annually, which was significantly smaller compared to 120kg were applied in the tea-annual land use systems. In this case, non-tea land use systems can be considered low-input farm systems while tea land use systems can be considered high input farms. The trend of application of organic inputs and household waste was similar to inorganic inputs with significant differences observed between wealthy classes and land use systems. In most cases, wealthy farms applied significantly more organic inputs than poor farm groups. Inorganic fertilizers were applied to crops with commercial orientation (Tea, banana, coffee and maize) and virtually absent to other crops (Table 5.6). Most organic inputs were applied on banana-annual land use system. The size of crop fields (Figure 5.3) was generally below 1 ha (varying between 0.01 – 1.2) but there were no significant differences between agricultural land use systems ($P< 0.5$).

5.3.3 Nutrient budgets

5.3.3.1 Plot level N Nutrient inputs/outputs, balances and stocks (production year 2010/2011) by land use systems

The mean contributions of the different N flows, nutrient balances and stocks at plot level are presented in Table 5.7a. All other fertilizer inputs exceeded inorganic fertilizers (IN1) mainly on food crops and not in the tea land use systems. The tendency was to apply more nutrient resources to permanent or commercial crops (tea, banana and coffee) than annual crops. Biological nitrogen fixation (IN4) was the third most import flow (9 kg $ha^{-1}yr^{-1}$) and organic fertilizers (15 kg $ha^{-1}yr^{-1}$). The level of sedimentation indicated soil erosion occurs within field. Crop harvest (OUT1) represented lesser (21 kg $ha^{-1}yr^{-1}$) nutrient export compared to (27 kg $ha^{-1}yr^{-1}$) by erosion (OUT5) and

(28 kg $ha^{-1}yr^{-1}$) leaching (OUT3). Leaching and erosion are therefore the largest N loss pathways. Nutrient export through OUT1 was generally higher in cereals and tea while nutrient export through crop residue removal (OUT2) was highest in napier grass, cereals and banana. N losses through leaching and gaseous losses/ denitrification were higher in tea and banana based crops which have higher nutrient application. Positive partial N balances were observed for tea and banana-coffee crops. This was mainly attributed to higher nutrient application in these crops. In contrast, low partial balances were observed for cereals and legume crops. Full N balance was negative for all crops, and more negative for napier grass and tea crops. N stocks were higher in tea and generally lower for other crops where low levels of fertilizers input are applied. The most nutrient extractive crop was napier grass > banana-cereal intercrops > cereals > tea.

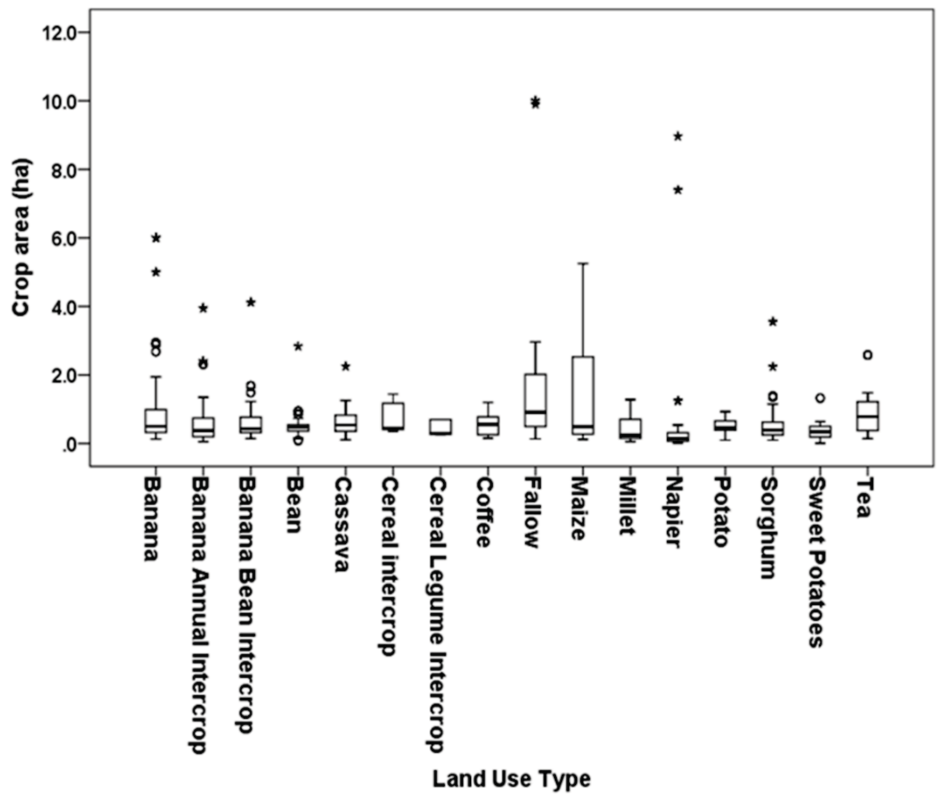

Figure 5.3: Crop area allocation, ranges, and outliers of different crops (production year 2010/2011, in kg ha^{-1} yr^{-1}). The box-and-whisker diagrams include: median value; (large rectangle) range of 50% of the samples; (cross bars) maximum and minimum values; (open circle) outliers; (stars) extreme values.

Table 5.5: Mean resource endowments belonging to the land use systems of South West Rwanda

Land use systems	Wealth class	No.	Total land area (ha)	persons per Household	Labour force	TLU	Inorganic fertilizers (kg ha^{-1})	Farm yard manure (FYM) (t ha^{-1})	Waste (t ha^{-1})
T	Poor	6	0.2(0.1)a	5(2)a	3(1)a	0.0(0)a	27(26)a	0.2(0.2)a	0.04(0.01)a
R1	Poor	5	0.1 (0.1)a	4(2)a	2(1)ab	0.0(0)a	6(1)a	0.1(0.0)a	0.03(0.04)a
R2	Poor	6	0.2(0.1)a	5(2)a	3(1)ab	0.0(0)a	0(0)a	0.1(0.2)a	0.05(0.10)a
R3	Poor	3	0.2(0.1)a	2(1)a	1(1)b	0.0(0)a	2(1)a	0.1(0.1)a	0.02(0)a
R4	Poor	3	0.2(0.1)a	5(3)a	2(2)ab	0.0(0)a	5(2)a	0.0(0.0)a	0.03(0.03)a
T	Medium	5	0.3(0.2)ab	8(2)b	4(1)a	0.1(0)b	75(46)b	0.1(0.2)b	0.05(0)b
R1	Medium	9	0.2(0.3)ab	5(2)b	2(1)ab	0.1(0.1)b	9(7)b	0.3(1.0)b	0.02(0.04)b
R2	Medium	14	0.3(0.1)ab	5(2)b	2(1)ab	0.1(0.1)b	0(0)b	0.2(0.2)b	0.02(0.03)b
R3	Medium	7	1.0(0.2)ab	6(3)b	2(1)b	0.1(0.1)b	0(0)b	0.3(0.7)b	0.03(0.03)b
R4	Medium	3	1.1(0.4)ab	6(1)b	3(1)ab	0.2(0.1)b	12(2)b	0.1(0.1)b	0.00(0.01)b
T	Rich	1	1.4(.2)bc	7(1)b	2(0)a	0.2(0.1)c	155(75)c	0.2(0.1)c	0.00(0)c
R1	Rich	7	0.4(0.3)bc	7(2)b	3(1)ab	0.2(0)c	11(10)c	0.3(0.2)c	0.06(0.05)c
R2	Rich	10	0.6(1.3)bc	5(1)b	2(1)ab	0.2(0.1)c	5(1)c	0.3(0.5)c	0.10(0.10)c
R3	Rich	9	0.8(0.7)bc	5(2)b	2(1)b	0.2(0.1)c	25(3)c	0.3(0.4)c	0.01(0)c
R4	Rich	3	1.3(0.7)bc	8(3)b	3(1)ab	0.3(0.2)	12(4)b	0.2(0.2)c	0.07(0.01)c
T	Mean (Total)	12	0.3(0.4)a	6(2)a	3(1)a	0.1(0.1)a	120(79)a	0.2(0.2)ab	0.040.01)a
R1	Mean (Total)	21	0.3(0.3)a	5(2)a	3(1)ab	0.1(0.1)b	10(8)b	0.3(0.7)cd	0.04(0.05)a
R2	Mean (Total)	30	0.4(0.7)a	5(2)a	2(1)ab	0.1(0.1)b	5(1)b	0.2(0.3)bc	0.04(0.08)a
R3	Mean (Total)	19	0.8(1.0)a	5(2)a	2(1)b	0.2(0.1)b	8(11)b	0.3(0.5)d	0.02(0.02)a
R4	Mean (Total)	9	0.9(0.7)a	6(3)a	3(1)ab	0.2(0.2)b	10(4)b	0.1(0.1)a	0.02(0.03)a

a,b,c; show significant differences between wealth groups (P<0.05). Values with similar letters are not significantly different

Table 5.6: Soil fertility management practices at crop level

Crop groups	Inorganic fertilizer (kg ha^{-1})	Farm yard manure (FYM) (t ha^{-1})	Waste (t ha^{-1})
Banana	15(8)a	0.2(0.2)abc	0.05(0.04) ab
Banana-Annual	4(8)a	0.3(0.8)c	0.02(0)ab
Banana-Bean	8(3)a	0.3(0.6)abc	0.04(0.02) ab
Bean	0(0)a	0.1(0.3)abc	0.02(0.03) b
Cassava	0(0)a	0.2(0.4)abc	0.08(0.02) ab
Coffee	14(12)a	0.3(0.4)bc	0.01(0)a
Fallow	0(0)a	0.1(0)abc	0.02(0)a
Legume-Cereal	0(0)a	0.1(0)ab	0.0(0)a
Maize	17(16)a	0.2(0.2)abc	0.0(0)a
Sorghum	0(0)a	0.1(0)abc	0.0(0)a
Millet	0(0)a	0.2(0.2)abc	0.0(0)a
Napier-grass	0(0)a	0.2(0.5)abc	0.02(0.01) ab
Potatoes	0(0)a	0.2(0.6)abc	0.05(0)ab
Sweet potatoes	0(0)a	0.1(0.1)abc	0.03(0.05) ab
Tea	131b	0.2(0.2)abc	0.0(0)a

[a,b,c;]show significant differences (P<0.05), Values with similar letters are not significantly different

5.3.3.2 Plot level P Nutrient inputs/outputs, balances and stocks (production year 2010/2011) by land use systems

Table 5.7b presents the different P flows, balances and stocks at plot level. Organic inputs (IN2) and sedimentation (IN5) contributed the largest P inputs (3 kg $ha^{-1}yr^{-1}$ and 5 kg $ha^{-1}yr^{-1}$, respectively) in mainly food crop production land use systems. IN1 contribution to P inflow in tea crop was significantly higher (P<0.05) due to higher application of inorganic fertilizers. Atmospheric deposition (IN3) was an important P inflow (6%) mainly in low input crops. More than 50% P (about 7 kg ha^{-1} yr^{-1}) in the fields is lost through erosion. P losses through harvest was higher in cereal and legume crops. P mining ranged between -1 to -13 kg $ha^{-1}yr^{-1}$ and was highest under cereal crops. The partial balance was positive for tea and banana based intercrops. P stocks were significantly higher (p<0.05) under tea crop compared to low input crops. P stock mining was observed under banana-cereals, cereals and legumes at rate of 1% yr^{-1}.

Table 5.7a. Plot level N Nutrient inputs/outputs, balances and stocks under different land use systems in South West Rwanda (production year 2010/2011, in kg $ha^{-1}yr^{-1}$). Standard deviations between parenthesis, N.A- not applicable

Crop group	Sample size	IN1N	IN2N	IN3N	IN4N	IN5N	OUT1N	OUT2N	OUT3N	OUT4N	OUT5N	N partial balance	N Full Balance	N-stocks ($kgha^{-1}$)	% of stock depleted yr^{-1}
Banana-Bean	21	2(6)a	15(24)ab	5(1)a	40(37)a	12(12)a	-25(22)ab	-14(33)ab	-37(11)a	-18(5)a	-19(11)a	-22(29)ab	-38(30)ab	4275(962)a	0.9
Banana-cereal	24	2(9)a	31(56)a	5(0)a	6(13)b	17(18)a	-25(47)ac	-14(19)ab	-42(16)a	-20(9)a	-24(18)a	-6(28)abc	-63(24)cde	4539(889)a	1.4
Banana-Coffee	62	1(5)a	18(22)ab	5(0)a	3(4)b	20(13)a	-6(11)bcd	-4(8)a	-35(10)a	-17(5)a	-25(13)a	10(19)cd	-39(19)ab	4838(1102)a	0.8
Cereal	78	1(5)a	20(30)ab	2(1)b	1(0)b	23(70)a	-41(35)	-8(15)ab	-14(5)a	-7(3)a	-29(49)a	-28(27)a	-52(39)bc	4509 (1013)a	1.2
Fallow	45	0(0)a	4(8)b	5(0)a	3(1)b	21(24)a	-1(6)d	-5(10)ab	-34(7)a	-16(4)a	-31(32)a	-2(12)bc	-53(20)bcd	4421(1042)a	1.2
Legume-Cereal	54	0(1)a	11(24)ab	2(0)b	48(55)a	14(13)a	-44(54)ab	-21(31)ac	-10(4)d	-5(2)d	-26(18)a	-53(63)e	-30(25)a	4068(634)a	0.7
Napier grass	90	0(0)a	9(31)ab	5(0)a	3(1)b	16(12)a	0	-36(42)c	-36(10)a	-18(6)a	-26(18)a	-27(29)a	-82(33)e	4565(1280)a	1.8
Root-Tuber	120	0(0)a	17(52)ab	4(2)c	2(1)b	20(21)a	-24(39)abc	-2(5)a	-24(17)c	-12(7)c	-28(19)a	-10(28)abc	-49(23)abc	4509(1339)a	1.1
Tea	18	59(83)b	15(27)ab	6(0)a	3(0)b	20(17)a	-39(44)a	-5(9)ab	-62(17)b	-34(9)b	-36(28)a	31(50)d	-73(44)de	6121(1597)b	1.2
Overall mean		3	15	4	9	19	-21	-12	-28	-14	-27	-16	-53	4649	1.1
% of total input/ output		5	30	8	18	38	21	12	27	14	26	N.A	N.A	N.A	N.A

5.3.3.3 Plot level K Nutrient inputs/outputs, balances and stocks (production year 2010/2011) by land use systems

The mean K flow, balances and stocks are presented in Table 5.7c. The largest inflows were from sedimentation (49 kg $ha^{-1}yr^{-1}$) and organic fertilizer applications (14 kg $ha^{-1}yr^{-1}$). Applications of manure, crop residue decomposition and burning were important K sources. Also more important than inorganic fertilizer application was atmospheric deposition (3 kg $ha^{-1}yr^{-1}$). Erosion contributes 67% of K field loss ranging between -29 to 78 kg $ha^{-1}yr^{-1}$. K losses through erosion were lower in tea crop compared to others. Crop residue removal of K was the most important in napier grass (-48 kg $ha^{-1}yr^{-1}$). Napier grass is harvested for livestock feeding. K losses through crop harvest were highest under root-tubers and legume-cereal intercrops. The full balance reveals that K mining was highest under Napier grass, legume, cereals and root tubers. K stocks were comparable among different land uses. Overall mining of K stocks was at rate of 2%. Higher mining occurred under Napier grass (3%), legume-cereals intercrop (3%), cereals (2%) and root-tubers (2%).

Table 5.7b. Plot level P Nutrient inputs/outputs, balances and stocks under different land use systems in South West Rwanda (production year 2010/2011, in kg ha-1yr-1). Standard deviations between parenthesis, N.A- not applicable

Crop group	Sample size	IN1P	IN2P	IN3P	IN5P	OUT1P	OUT2P	OUT5P	P partial balance	P Full Balance	P-stocks (kgha^{-1})	% of stock depleted yr^{-1}
Banana-Bean	21	1(5)a	2(5)a	1(0.1)a	3(3)a	-3(5)a	-1(3)a	-5(3)a	0(7)a	-1(7)ab	866(253)a	0
Banana-cereal	24	1(7)a	8(18)a	1(0)a	4(4)a	-9(18)b	-6(8)b	-6(4)a	-5(11)ab	-6(11)abc	878(243)a	1
Banana-Coffee	62	1(3)a	5(6)a	1(0.1)a	6(5)a	-1(2)a	-1(1)a	-7(5)a	4(7)a	4(8)a	1080(347)a	0
Cereal	78	1(3)a	5(9)a	0.4(0.1)b	6(12)a	-14(12)c	-3(5)c	-8(9)a	-11(12)b	-13(13)c	927(254)a	1
Fallow	45	0(0)a	1(2)a	1(0)a	5(4)a	0(2)b	-1(1)a	-8(6)a	0(3)a	-2(5)ab	895(286)a	0
Legume-Cereal	54	0(1)a	2(3)a	0.3b	4(4)a	-4(7)ab	-2(3)ab	-7(5)a	-5(7)ab	-8(8)bc	821(116)a	1
Napier grass	90	0(0)a	2(9)a	1(0)a	4(4)a	0(2)a	-4(5)a	-7(4)a	-2(8)ab	-4(8)abc	930(300)a	0
Root-Tuber	120	0(0)a	4(16)a	1(0.2)c	5(5)a	-4(6)ab	-1(2)ab	-8(6)a	-1(13)a	-2(14)ab	922(342)a	0
Tea	18	40(58)a	4(11)b	1(0)b	5(4)b	-4(5)ab	-1(1)ab	-9(7)a	40(53)c	37(54)d	1204(423)b	-3
Overall mean		2	3	1	5	-4	-2	-7	-1	-3	974	0
Percentage of total input/ output		16	32	6	45	33	14	53	N.A	N.A	N.A	N.A

Table 5.7c. Plot level K Nutrient inputs/outputs, balances and stocks under different land use systems in South West Rwanda (production year 2010/2011, in kg $ha^{-1}yr^{-1}$). Standard deviations between parenthesis, N.A- not applicable

Crop group	Sample size	IN1K	IN2K	IN3K	IN5K	OUT1K	OUT2K	OUT3K	OUT5K	K partial balance	K Full balance	K-stocks	% of stock depleted yr^{-1}
Banana-Bean	21	1(3)a	11(16)a	4(1)a	46(54)a	-14(13)abc	-11(21)a	-0.2(0.3)ab	-58(46)ab	-13(24)abc	-21(28)ab	1930(186)ab	1
Banana-cereal	24	2(2)a	32(57)a	4(0)a	57(72)a	-7(13)ab	-45(62)bc	-0.3(0.5)abc	-74(75)ab	-18(64)abc	-31(60)abc	1954(217)b	2
Banana-Coffee	62	2(2)a	18(25)a	4(0)a	68(77)a	-3(5)a	-9(15)a	-0.2(0.2)ab	-78(74)a	8(23)ad	1(30)a	1943(181)b	0
Cereal	78	1(3)a	15(27)b	1(0)c	48(56)a	-11(9)ab	-21(46)ab	-0.1(0.1)a	-72(72)ab	-16(42)abc	-39(51)bc	1984(199)b	2
Fallow	45	0(0)a	5(12)a	4(0)a	50(76)a	-1(4)a	-7(14)a	-0.1(0.2)a	-69(85)ab	-2(15)ab	-17(26)ab	1852(182)ab	1
Legume-Cereal	54	0(1)a	8(17)a	1(0)c	37(60)a	-27(32)bc	-14(20)a	-0.1(0.1)a	-61(70)ab	-32(35)c	-55(40)c	1925(152)ab	3
Napier grass	90	0(0)a	9(34)a	4(0)a	50(72)a	0(0)ac	-48(56)c	-0.5(0.6)c	-71(75)ab	-38(37)c	-56(43)c	1901(193)ab	3
Root-Tuber	120	0(0)a	15(52)a	2(1)b	46(54)a	-34(56)c	-3(11)a	-0.1(0.4)a	-59(54)ab	-22(49)bc	-33(56)bc	1862(171)ab	2
Tea	18	41(68)d	17(43)a	4(0)a	23(40)a	-15(17)abc	-7(12)a	-0.4(0.6)bc	-29(33)c	36(63)d	34(59)d	1912(155)ab	-2
Overall mean		2	14	3	49	-15	-18	-0.2	-66	-17	-32	1918	2
Percentage of total input/ output		3	20	4	73	15	18	0.2	67	N.A	N.A	N.A	N.A

5.3.4 Farm level nutrient input/outputs, balances and stocks (production year 2010/2011) by land use systems

Nutrient flows and balances per farm group are presented in Table 5.8. Nutrient imports into the farms were highly variable between high input farms (tea farms) and low input farm groups (R1, R2, R3 and R4). Significantly ($P<0.05$) higher import of inorganic fertilizers (IN1) was recorded in tea farms compared to low input farms. Although IN2 and IN3 were in limited amounts, they generally exceeded IN1 in non-tea farm groups. Biological N fixation (IN4) was more important input than IN1, IN2 and IN3 in non-tea farms. OUT1 was highest in T and R1 farms that are commercially oriented. Nutrient export out the farms through crop residue removal was negligible and was used to feed livestock and mulch within the farm. Crop produce leaving the farm (OUT1) constitute about 1 kg N $ha^{-1}yr^{-1}$, 0 kg P $ha^{-1}yr^{-1}$ and 1 kg K $ha^{-1}yr^{-1}$. Leaching (OUT3) was highest N loss pathway 36 kg N $ha^{-1}yr^{-1}$and only 4 kg K $ha^{-1}yr^{-1}$. Erosion (OUT5) contributed the highest loss for P (8 kg P $ha^{-1}yr^{-1}$), 71 kg K $ha^{-1}yr^{-1}$ and 29 kg N $ha^{-1}yr^{-1}$.

Overall, N and K partial balance were positive for only T, R3 and R4 (Table 5.9). P partial balance was positive across all farms. Partial balance for tea farms was significantly different ($P<0.05$) than others. Apparently, most farmers export (OUT1 and OUT2) fewer nutrients than they import (IN1and IN2). The full balance for low input farms (R1, R2, R3 and R4) that are food based were generally negative reflecting the effect of low nutrient imports and high field nutrient losses due to leaching, erosion and denitrification on farms. Tea farms that are intensively managed and commercial based import more inorganic fertilizers resulting in a positive P and K balance. It was observed that N full balances were negative across farms and were not significantly different across farms ($P< 0.05$). During the monitoring period, overall mining was about 1% of the total N stocks, close to 0% of P stocks and 0.6% of K stocks. High stocks mining occurred in low input farms.

5.3.5 Farm nutrient balances by wealth group and landscape position

We calculated full nutrient balances between different wealth classes (Figure 5.4) and also for different slope positions (Table 5.10). There was no significant differences ($P<0.05$) in the nutrient balances between family wealth classes at farm level. On average the P balance was positive for all farms and higher in medium and poor family wealth class farms compared to the rich class. N and K balances were highly negative in rich farms and slightly lower in the poor. Apparently rich farms are depleting soils. We did not find significant differences ($P<0.05$) between slope positions although more nutrient depletion occurred in the mid-slope than in the upper slope and foot slope (Table 5.10). N and K balance was generally negative at all slope positions while P balance was positive.

Table 5.8. Farm nutrient input/outputs under different land use systems in South West Rwanda (production year 2010/2011, in kg ha^{-1} yr^{-1})

	N					P					K				
Crop systems	T	R1	R2	R3	R4	T	R1	R2	R3	R4	T	R1	R2	R3	R4
IN1	30(31)a	2 (3)b	1(2)b	1(2)b	3(6)b	11(7)a	1(2)b	0(1)b	0(1)b	1(2)b	17(13)a	2(3)b	1(1)b	0(1)b	1(2)b
IN2	13(15)a	7(15)a	8(10)a	11(14)a	9(21)a	3(3)a	1(2)a	1(2)a	3(4)a	2(4)a	16(20)a	9(20)a	7(9)a	13(17)a	8(15)a
IN3	6(3)a	5(1)a	4(1)a	5(2)a	5(1)a	0.9(0.5)a	0.8(0.2)a	0.7(0.2)a	0.8(0.3)a	0.8(0.2)a	4(2)a	3(1)a	3(1)a	3(1)a	3(1)a
IN4	19(14)a	19(20)a	11(26)a	24(35)a	4(3)a	0	0	0	0	0	0	0	0	0	0
IN5	28(25)a	15(8)a	16(9)a	19(7)a	22(5)a	6(6)a	4(4)a	5(2)a	5(2)a	6(2)a	52(28)a	46(22)a	48(32)a	64(35)a	41(23)a
OUT1	-12(7)a	-10(11)a	-9(12)a	-6(12)a	-6(7)a	-2(1)ab	-3(4)a	-2(2)ab	-1(1)ab	0(1)b	-6(4)	-7(9)	-6(8)	-4(8)	-1(4)
OUT2	0(0)a	0(0)a	-1(3)a	-1(4)a	0(0)a	0(0)a	0(0.1)a	-0.4(1.1)a	-0.1(0.4)a	0(0)a	0(0)	-0.5(0.9)	-2.8(8.2)	-0.2(0.6)	-0.1(0.1)
OUT3N	-48(18)a	-35(15)ab	-23(9)b	-39(22)a	-33(16)ab	0	0	0	0	0	-5(7)a	-5(6)a	-4(3)a	-4(5)a	-1(2)a
OUT4N	-30(13)a	-23(9)ab	-13(4)c	-20(9)bc	-17(7)bc	0	0	0	0	0	0	0	0	0	0
OUT5N	-36(24)	-24(10)	-23(9)	-25(9)	-35(7)	-8(6)a	-6(3)a	-8(3)a	-7(2)a	-9(3)a	-77(40)a	-70(26)a	-69(31)a	-83(35)a	-57(25)a

Standard deviations between parenthesis; [a,b,c] show significant differences ($P<0.05$). Values with similar letters are not significantly different

Table 5.9. Farm level nutrient balances, Stocks and annual depletion rate in the 0 – 20 cm topsoil

	N						P						K					
Crop systems	T	R1	R2	R3	R4	Overall N mean	T	R1	R2	R3	R4	Overall p mean	T	R1	R2	R3	R4	Overall K mean
Partial Nutrient Balance	32 (31)a	-1 (18)b	-1 (16)b	5 (18)b	6 (16)b	8	12 (5)a	0 (4)b	0 (3)b	2 (3)b	3 (4)b	3	30 (30)a	4 (23)b	-1 (12)b	9 (18)b	8 (13)b	8
Full nutrient Balance	-30 (24)a	-45 (24)a	-28 (22)a	-31 (16)a	-48 (13)a	-36	13 (13)a	-1 (8)b	-2 (5)b	1 (5)b	1 (6)b	2	2 (28)a	-18 (26)ab	-23 (18)b	-10 (23)ab	-8 (12)ab	-8
Nutrient stocks (kgha-1)	5897 (1551)a	4680 (1038)b	3802 (901)c	4182 (812)c	4684 (1109)b	4594 (1328)	1250 (352)a	1091 (246)b	838 (256)c	726 (225)d	853 (218)c	959 (330)	1857 (176)a	1880 (210)a	1985 (216)b	1925 (156)ab	1888 (136)a	1914 (194)
% of stock depleted yr^{-1}	0.5	1.0	0.8	0.7	1.0	1	-1.1	0.1	0.2	0.1	0.1	0.2	-0.1	1.0	1.2	0.5	0.4	-1

Table 5.10. Farm level full nutrient balance at difference slope positions

Slope position	Bottom					Midslope					Upslope				
Crop systems	T	R1	R2	R3	R4	T	R1	R2	R3	R4	T	R1	R2	R3	R4
Sample size	2	3	3	3	2	4	5	9	6	4	6	13	18	10	3
N-Balance	- 26(0)a	-55(42)a	-22(33)a	-13(6)a	-43(0)a	-25(31)a	-51(20)a	-32(15)a	-30(15)a	-50(15)a	-33(25)a	-40(21)a	-27(25)a	-37(15)a	-47(14)a
P-Balance	12(2)a	0(15)a	-4(4)a	0(6)a	12(0)a	18(8)a	1(6)a	-2(4)a	4(6)a	0(4)a	12(16)a	-2(8)a	-2(5)a	0(3)a	0(6)a
K-balance	1(0)a	-18(19)a	2(5)a	3(15)a	6(0)a	27(42)a	-18(32)a	-20(18)a	-12(31)a	-15(10)a	-8(17)a	-17(27)a	-29(16)a	-12(21)a	-1(8)a
Overall N Balance			-30 (28)a					-37 (20)a					-35 (22)a		
Overall P Balance			2 (10)a					3 (8)a					1 (10)a		
Overall K Balance			-3 (14)a					-11 (29)a					-17 (22)a		

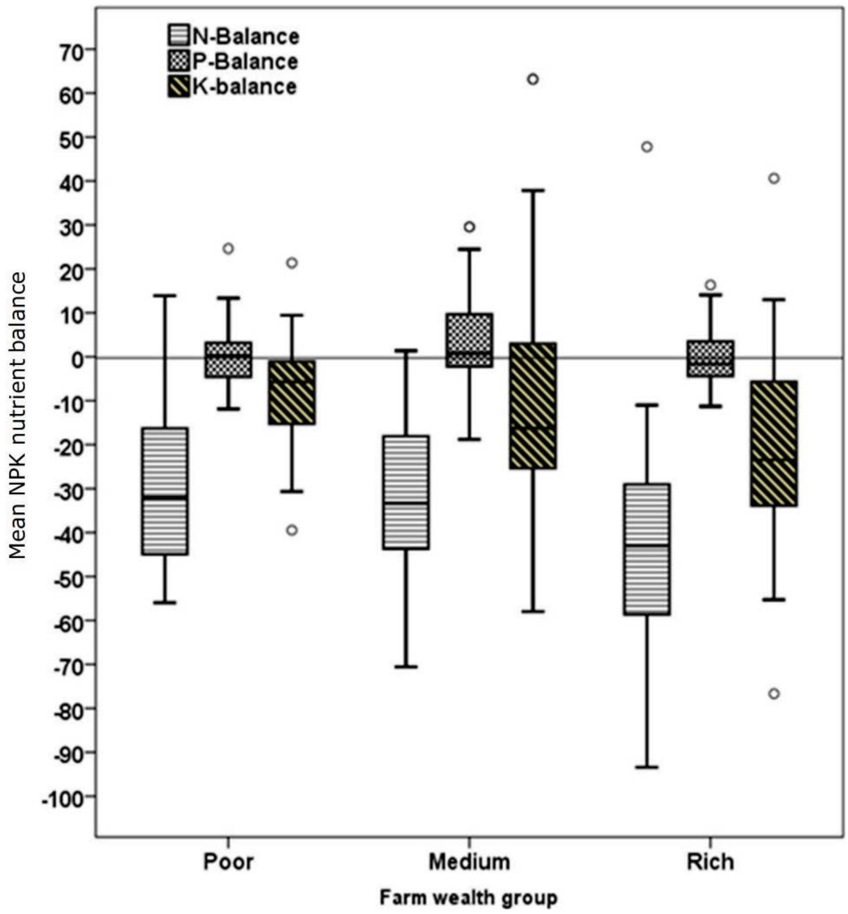

Figure 5.4: Full nutrient balances on study farms in different wealth groups

5.4 Discussion

5.4.1 Soil nutrient concentrations, mean nutrient stocks and soil fertility management

The soil nutrient analysis indicated low to deficient levels of SOC, N and K, while P concentration was above the soil fertility critical level for crop production according to the thresholds reported in Foster (1971), Gourley (1999) and Okalebo et al. (2002). Therefore, N and K are limiting nutrients for crop production in soils of south-west Rwandan land use system. The nutrient status in agricultural soils indicates on going nutrient depletion in agricultural land use systems. Organic matter related parameters like N and P contents were higher in the tea-annual land use system and low in others that are mainly food crop land use (Table 5.4). Fertilizer and manure inputs have been low on mainly food crop land use systems. Low levels of nutrients in agricultural soils clearly points to the problem of sustainable food production and food security in Rwanda. Agricultural soils cultivated without adequate nutrient replenishment cannot reach their full crop production potential and are at risk of irreversible degradation (Lal 2010). Without maintaining adequate soil fertility levels, crop yields cannot be sustained, increase over time, or respond to improved agricultural management practices (van der Velde et al. 2013). Large amounts of inputs will be required to build soil SOC capital and restore soil fertility and improve crop productivity. Rich farmers applied significantly more inorganic fertilizers than poor farm groups which was similar to findings in Ethiopia (Elias et al., 1998 and Haileslassie et al. 2006, 2007). Food based land use systems had higher exchangeable K values than the tea-annual land use systems. Kanmegne (2004) has explained that higher exchangeable K values were probably because of crop residue burning practices during seed bed preparation, which is also a practice in Rwanda. On average, 7 kg of inorganic fertilizers was applied by low input farms annually which is slight low than average annual application of 8 -15 kgha^{-1} in Africa and compares very poorly to an average global value of 90 kgha^{-1} (Bekunda et al. 2010).

5.4.2 Plot level Nutrient budgets of land use systems

Currently, high nutrient mining occurs under food crops land use systems compared to banana-coffee and tea land use systems that are intensively managed and have commercial orientation. Similar findings were reported Burkina Faso (Lesschen et al. 2007). The trend of N mining was highest for napier grass > tea > banana-cereal > fallow > cereals > root – tuber > banana – coffee > banana – bean > legume – banana, in declining order, respectively. A similar situation of high nutrient mining under napier grass is reported by Van den Bosch et al. (1998) where considerable amounts of FYM (IN2) were applied to Napier grass fields, but the amounts of N and K returned to the fields via manure were usually less than the amounts removed with the grass to feed livestock. Compared to our calculations, similar rates of crop nutrient losses are reported in Central Highlands of Ethiopia (Haileslassie et al. 2007) as; -46kg N ha^{-1} yr^{-1}, 4 kg P ha^{-1} yr^{-1} and -57 kg K ha^{-1} yr^{-1} and in the slash and burn agriculture in southern Cameroon (Kanmegne 2004) -67 kg N ha^{-1} yr^{-1}, -1 kg P ha^{-1} yr^{-1} and -4 kg K ha^{-1} yr^{-1} for plots under conventional fertility management and -63 kg N ha^{-1} yr^{-1}, 0 kg P ha^{-1} yr^{-1} and 1 kg K ha^{-1} yr^{-1} for plots where Kitchen residues and animal manure was recycled. P and K balances were positive for banana-coffee and tea because these crops receive relatively high organic inputs that could supply P and K. Tea has high application of inorganic fertilizers. Our findings are also consistent with studies in East Africa (Embu, Kenya) by De Jager et al. (1998) and Van den Bosch et al. (1998) where considerable amounts of mineral nutrients were applied to cash crops and very little to staple food crops. It has also been reported that banana and coffee crop in the Southern

Nation Nationalities Peoples Region (SNNPR) of Ethiopia received the highest amount of organic inputs (IN2) per unit area (Elias et al. 1998; Tsegaye, 2001; Haileslassie et al. 2005). Wortmann and Kaizzi, 1998 also found that nutrient balances for banana were near zero because the banana-based land use type receive crop residue from other land use types.

Leaching and erosion are the most prominent nutrient loss pathways among land use systems compared to crop harvest and denitrifications. N leaching and denitrification fluxes were -28 kg N $ha^{-1}yr^{-1}$ and 14 N $ha^{-1}yr^{-1}$, rates which is close to the result of Van den Bosch et al. (1998) and Haileslassie et al. (2005). Soil loss estimates in annual and perennial land uses was between 0.1 to 100 ton ha^{-1} yr^{-1} with average erosion in annual crops, banana-coffee and tea at; 17, 12 and 6 t ha^{-1} yr^{-1} (Wasige et al. 2013c) corresponding to nutrient losses that varied bewteen -19 to -36 kg N $ha^{-1}yr^{-1}$, -5 to -9 P $ha^{-1}yr^{-1}$, -29 to -78 kg K $ha^{-1}yr^{-1}$. De Jager et al. (2001) in Kenya and Haileslassie et al. (2005) in Ethiopia reported that erosion was the most import nutrient loss pathway in crop field ,contributing total nutrient balance removed from cereal land use of about 70% of N, 80% P and 63% of K

5.4.3 Farm nutrient budgets

Biological nitrogen fixation, organic fertilizers and atmospheric deposition were the most important inputs on low input land use systems and exceed IN1. Inorganic fertilizer use was as low as less than 2 kg N $ha^{-1}yr^{-1}$ in the low input land use systems (R1, R2, R3 and R4) and about 30 kg N $ha^{-1}yr^{-1}$, 11 kg P $ha^{-1}yr^{-1}$, 17 kg K $ha^{-1}yr^{-1}$ in tea farms (T) compared to about 10.8 kg N $ha^{-1}yr^{-1}$ in south western Uganda (Bekunda and Manzi 2003) and 6 to 29 kg N $ha^{-1}yr^{-1}$ in Kenya (De Jager et al. 1998, 2001) and about 8.5 kg N $ha^{-1}yr^{-1}$ for Ethiopia (Haileslassie et al. 2005).

The partial farm nutrient balances were marginally negative under food crops production oriented farms that were mainly are of low fertilizer input and highly positive under intensively managed tea land use system. More positive partial nutrient balances were reported by Van den Bosch et al. (1998) in three different districts in Kenya where farmers applied more inputs with average partial balances of 34 kg N ha^{-1} yr^{-1}, 12 kg P ha^{-1} yr^{-1} and 27 kg K ha^{-1} yr^{-1}. The partial balances were lower for Central Highlands of Ethiopia (Haileslassie et al. 2007) with an average of -19 kg N ha^{-1} yr^{-1}, 3 kg P ha^{-1} yr^{-1} and -25 kg K $ha^{-1}yr^{-1}$ indicating higher nutrient depletion. Also, De Jager et al. (2001) in Kenya reports lower partial balances between conventional and low-external input agriculture (LEIA) farm management at an average of -58 kg N ha^{-1} yr^{-1} and -6 kg P ha^{-1} yr^{-1} indicating high levels of nutrient mining. While partial balances were highly variable in Murewa, northeast Zimbabwe where N and P partial balances were largest on the wealthy farms (−20 to

+80 kg N per farm and 15–30 kg P per farm) due to high nutrient applications (Zingore et al. 2007).

The full farm nutrient balances were negative for N and K balance and positive for P. Probably N and K are more subjected to more losses such leaching and/ or volatilization that are not transmission avenues for P. The trend of NPK nutrient balance in this study is comparable with 57 peer-reviewed studies carried out in Africa (Figure 5.5) by (Cobo et al. 2010). The analysis of 57 studies confirmed the expected trend of negative balances in the continent for nitrogen and potassium, where >75% of selected studies had mean values below zero. For phosphorus only 56% of studies showed negative mean balances. From this study, it was revealed that despite the overall negative trend on nutrient balances in Africa, positive balances could also be found on the continent and therefore soil nutrient mining cannot be generalized across the continent. This is evidenced in Figure 5.5, especially for P and where mean values from 44, 24 and 15% of the studies (for P, N and K, respectively) were above zero, as well as in all positive observations from many of the studies. The Differences in nutrient balances among systems, system components, sites and seasons in the 57 studies was attributed to a great diversity of factors, which typically depend on the spatial scale of the study. We believe another reason for the differences of farm nutrient balances results across different locations in Africa could be brought about by the level of farm commercial production orientation. For example, a high negative nutrient balance of the farms studied by De Jager et al. (2001) and Van den Bosch et al. (1998) Kenya could be due to large proportion of the farm produce sold to market since farmers in this region are more commercially oriented.

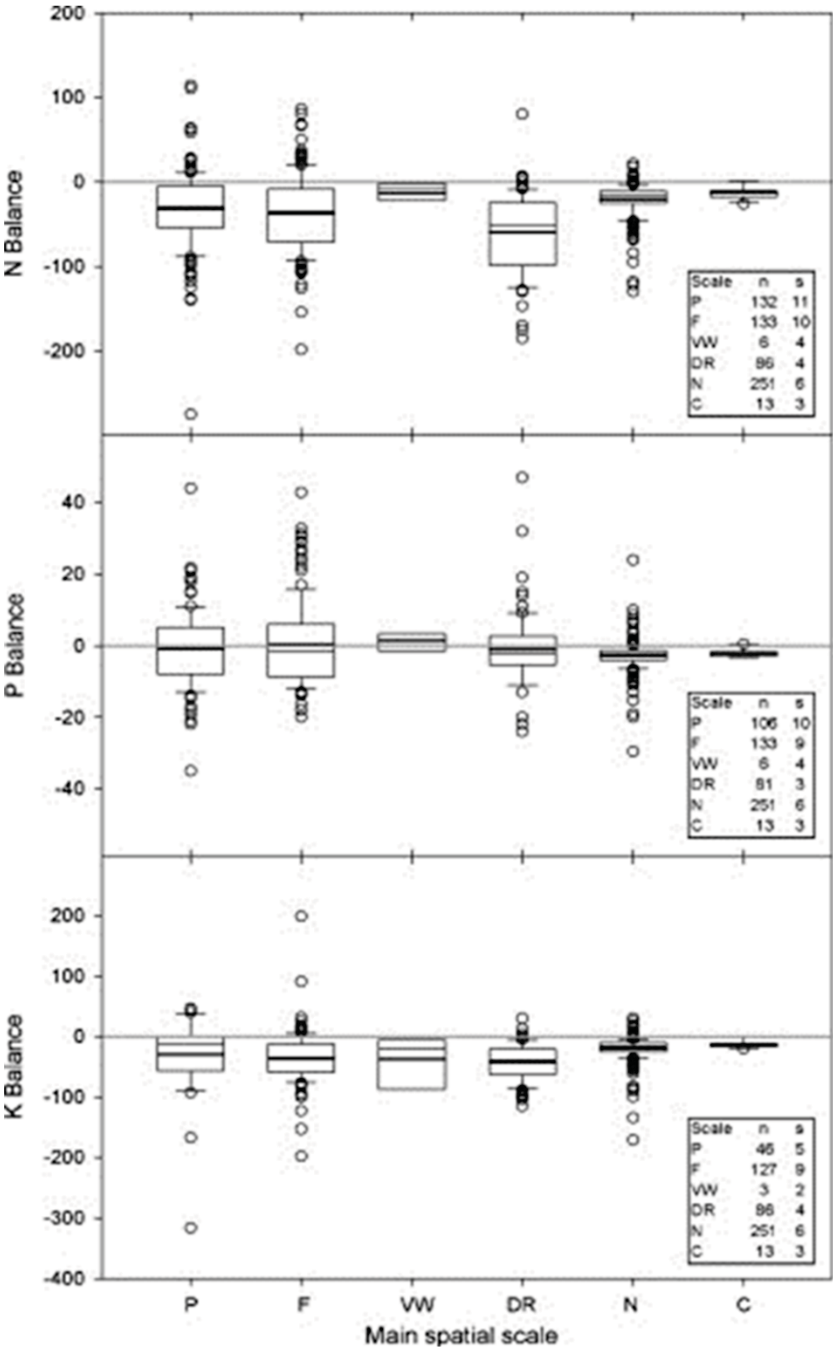

Figure 5.5: Nutrient balances at different spatial scales in Africa (Cobo et al., 2010). P: plot, F: farm, VW: village and watershed, DR: district and region, N: nation, C: continent. Only data expressed as kg ha^{-1} $year^{-1}$ and derived from full nutrient balances studies were plotted for the comparison. Number of observations (n) and studies (s) per category are shown in the rectangles.

A comparison of farm and plot scale nutrient balances, there were higher nutrient depletion at plot level than indicated by farm level. This is because, at the farm scale, there nutrient pools/ reserves e.g., redistribution units (RU), secondary production units (livestock units) that are mainly of manure type and stocks in the store that reduce the negative balance. Haileslassie et al. (2007) also reported this trend in the central highlands of Ethiopia. They explained that aggregation of nutrient balances from the field to the farm level may also mask relations between individual farming activities or management strategies and access to resources. For example fields with strongly negative nutrient balances within a farm may be masked by fields' 'with a balanced or positive nutrient budget. The latter may be caused either by within-farm nutrient redistribution (e.g. via manure application) or via different degrees of external inputs such as mineral fertilizer. Also, farm nutrient balances of rich farmers were highly negative than the medium and poor farmer. Similar findings were reported by Elias et al. (1998) and Haileslassie et al. (2007) in Ethiopia who explained that rich farmers tend to export more farm products to the market than nutrient returned to the farm. In relation to the nutrient stocks the results are less alarming for all nutrients with high depletion rates of about 1% for N. The rate of 1% annual loss of N stocks per year was lower than 3.5% observed by Haileslassie et al. (2007) in the central highlands of Ethiopia.

In the light of the nutrient loss pathway, low level of nutrients are removed through crop harvest or organic materials compared to nutrient loses through the so so-called 'hard-to quantify' flows (i.e., leaching, gaseous losses and erosion). N loss of about (36 N ha^{-1} yr^{-1}) was by OUT3 (leaching), OUT5 (erosion) N (29 kg ha^{-1}yr^{-1}), OUT4 (gaseous loses) contributed about (21 kg N ha^{-1} yr^{-1}). Only 9 kg N ha^{-1} yr^{-1} was lost through crop harvest. Van den Bosch et al. (1998) reported level of leaching for all farms was 53 kg N ha^{-1} yr^{-1}, with large variations between farms, ranging from 9-123 kg N ha^{-1} yr^{-1}. Gaseous losses were calculated at 24 kg N ha^{-1} yr^{-1}. Erosion contributed the highest loss for P (8 kg ha^{-1}yr^{-1}), K (71 kg ha^{-1}yr^{-1}) (Figure 5.4). Henao and Baanante (2006) have reported that nutrient losses due to erosion range from of 10 to 45 kg of NPK ha^{-1} yr^{-1} in Africa. High nutrient losses are reported by Haileslassie et al. (2005) in Ethiopia (81 kg N ha^{-1}yr^{-1}, 12 kg P ha^{-1}yr^{-1} and 48 kg K ha^{-1}yr^{-1}). The differences could accrue from the scales at which soil erosion is estimated. The most important improvement in our study was the application of a new spatially explicit hydrological model by Wasige at al. (2013c) for the feedback between erosion and sedimentation at pixel level (30 m x 30 m). This resolution is in the order of agricultural field size (0.01 – 1.2 ha) in south west Rwanda which allowed us to couple the model to the farm nutrient balance research. This is so far the only model that can simulate erosion and sedimentation in a daily continuous way and in spatially explicit quantitative terms with minimum dataset at this scale.

Erosion estimates for the nutrient balance study by Haileslassie et al. (2005) does not report the scale which LAPSUS (LandscApe ProcessS modeling at mUltidimensions and Scales) model was applied. Upto 1 km grid cell resolution for LAPSUS model was applied to estimate sedimentation and erosion in the Lesschen et al. (2007) nutrient balance study. In many cases estimate of soil erosion rates at large scale units exaggerates soil loss because of inappropriate extrapolation and generalization (Stroosnijder, 2012). This means that topography, sedimentation and erosion are not correctly represented at plot, farm and catchment scale, because small valleys and ridges are leveled out (Temme et al. 2006). Nutrient gains and losses through sedimentation and soil loss assessment at smallholder farm plots of the scale in south west Rwanda needs a sediment balance model with ins' and outs' per grid cell at high spatial resolution. Some studies rarely considered sedimentation or erosion in the full nutrient balance calculation (e.g., Wortmann and Kaizzi 1998; Ebanyat et al. 2010) despite it being a substantial process in the system. Other studies have estimated nutrient losses as an input in nutrient balance studies on the basis of large homogenous units that can be as large as sub watershed to catchment or regional scale using an empirical model (USLE) (e.g., Stoorvogel et al. 1993; De Jager et al. 1998, 2001; Olupot et al. 2004). Empirical models are based on the famous USLE (Wischmeier and Smith, 1978). Although results have been widely published, the USLE does not calculate soil loss but sediment production only because it has no equations or algorithms to calculate transport and sedimentation processes. Thus it only works in situations for which it was conceived, sloping fields or parts of hillslopes that have little deposition, only erosion. It can only be used on a catchment scale when combined with a catchment based sediment delivery ratio system (Wasige et al. 2013c).

Biological N-fixation in crops was an important source of N with estimated ranges from 4 to 24 kg ha^{-1} yr^{-1}. Amounts of N fixed up to 450 kg ha^{-1} yr^{-1} have been recorded in the tropics (Smithson and Giller 2002). Haileslassie et al. (2005) reported 18 kg N ha^{-1} yr^{-1} for pulses. Atmospheric deposition (IN3) contributed about 4.7, 0.8, 3 kg kg ha^{-1}yr^{-1} of N, P and K deposition. Globally, N deposition rates are estimated to be 5 kg N ha^{-1} for less densely populated and non-industrial countries and range from 20 kg N ha^{-1} yr^{-1} to 50 kg N ha^{-1} yr^{-1} in countries of Western Europe and parts of China (Sheldrick et al. 2003). Mengel and Kirkby (1996) also discussed that wet deposition for N can be as high as 60 kg ha^{-1} yr^{-1} depending on proximity to a city and amount of precipitation. Haileslassie et al. (2005) reported that the humid regions of Gambela, receive relatively high nutrient inputs via IN3 (5.3 kg N ha^{-1} yr^{-1}, 0.9 kg P ha^{-1} yr^{-1} and 3.5 kg K ha^{-1} yr^{-1}), while the drier regions (Somali and Afar) receive lower quantities (3.2 kg N ha^{-1} yr^{-1}, 0.5 kg P ha^{-1} yr^{-1} and 2.1 kg ha^{-1} yr^{-1}).

There were no significant differences for N, P and K nutrient balances at all slope positions. This finding is similar to results of Nabahungu (2012) in central plateau of Rwanda where there was no significant difference in N and P balances between hillsides and the wetlands. In our study, high N and K nutrient mining was mostly occurred in mid slope and valley bottom positions. Haileslassie et al. (2007) in the central highlands of Ethiopia where the magnitude of nutrient depletion on the foot slope was low compared with the plots on the mid slope and upslope. Thapa and Paudel, (2002) also reported that hillslope paddy lands of two mountain watersheds in the western hills of Nepal suffer from N and K deficiency compared to upland crop terraces and Valley paddy land. Similarly, a net negative balance of nitrogen (N) was reported on all types of lands and potassium (K) was negatively balanced on valley and hillslope paddy fields, and positively on upland crop.

5.5 Conclusion

The mean farm partial balance was marginally positive, indicating to the limited nutrients applied in soil fertility management. However, full soil nutrient balances results confirmed soil fertility depletion. N and K overall mean full balances were generally negative across farm land use systems while P was positive. The trend of N mining was highest for napier grass > tea > banana-cereal > fallow > cereals > root – tuber > banana – coffee > banana – bean > legume – banana, in declining order, respectively. A positive P and K balance was only observed in banana – coffee and tea crops that are intensively managed commercial crops and mining occurred in food crops land use systems. It was observed that stocks in tea land use system were increasing at rate of 3 %. Nutrient losses were mainly the result of the "hard to quantify" nutrient loss pathways (i.e. erosion, denitrification and leaching). Water erosion influences the soil nutrient balance very much in south west Rwanda, since the erosion rates were high, because of high rainfall and complex hilly topography. Erosion contributed the highest loss for P and K, while N losses were highest by leaching. Generally, there is nutrient mining in south west Rwanda agricultural systems since soil nutrient stock has to offset the negative balances each year. In relation to the nutrient stocks the results are less alarming for all nutrients with depletion rates of about 1% per year. This study highlights the need to consider soil fertility restoration through integrated soil fertility management to improve nutrient capital on smallholder farms for sustainable food production in south west Rwanda.

5.6 Acknowledgements

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Chapter 6

General discussion

6.1 Introduction

The upstream catchments of the Lake Victoria Basin (LVB), such as the Kagera basin studied here, have undergone large land use and land cover changes in the last 100 years. Forests have changed to agricultural land with annual crops and permanent crops such as tea, coffee and banana. On the one hand this has generally increased the agricultural productivity of the area, on the other there is evidence for environmental degradation, with soil degradation (i.e. loss of organic matter and nutrients), increased soil erosion from the farms and increased sediment and nutrient loads in the river systems. This is potentially threatening Lake Victoria basin's ecological quality and productivity (in terms of agricultural production, fishing stocks and transport). The basin environmental problems include: soil erosion, biodiversity loss, water quality decline, fluctuating Lake water levels, eutrophication, and proliferation of invasive species. This thesis attempts to link long term land use and change (LUCC) to farm soil quality changes, and then integrate sediment and nutrient losses from farm scale to catchment scale with a spatially explicit model. Its overall goal is to contribute to understanding of how land use and land cover change affect hydrology, sediment fluxes and soil fertility as a scientific basis for evaluation of the dynamics of sediment/ nutrient loading and eutrophication in Lake Victoria.
In this chapter the various elements of this research are summarized and brought together to answer the following research questions:

1) What are the land use changes in the last 100 years?
2) How has this affected the carbon stocks?
3) What are the current erosion patterns and catchment soil loss rates?
4) What is the simulated change in sediment dynamics from 1974 until now as a result of land use change?
5) What is the nutrient balance of the different farming systems and what role does erosion play in this balance?
6) What are the implications for sustainable land management?

6.2 Spatially explicit long-term assessment of land use and land cover changes (LUCC)

Land use and land cover change (LUCC) is considered one of the predominant forces that affects land and water degradation. Lack of consistent long-term land use and land cover change data was always considered a critical gap in our knowledge of the role of LUCC in soil degradation and soil erosion in the LVB. A combination of ancillary data, colonial maps and satellite imagery (Landsat) were interpreted to construct and present spatially explicit LUCC dynamics for the last century in Kagera basin, the upper catchment of LVB. The results showed that large trends of LUCC have rapidly occurred over the last 100 years. The most dominant LUCC processes were gains in farmland areas (60%) and a net reduction in dense forest (7% to 2.6%), woodlands

(51% to 6.9%) and savannas (35% to 19.6%) between 1901 and 2010. The overall farmland expansion rate of 0.57% yr^{-1} (calculated over the whole period 1901-2010) is not as high as reported in other studies for Sub Saharan Africa but at certain periods in the studied time frame higher rates were observed (1.56 % yr^{-1} between 1974 and 1985). The results of this study offer important directions for follow up research aimed at quantifying impacts of LUCC on depletion of SOC (chapter **3**) and nutrient stocks (chapter **5**), and increase in sediment and nutrient loading in the river (chapter **4**).

6.3 Land use change impact on soil organic carbon stocks

Contemporary land use significantly drives soil organic carbon (SOC) stock variation and the global carbon cycle. Given the high level of land cover change that has happened (chapter **2**), research on how changes in LUCC impact on soil C stocks in the Lake Victoria landscapes were profoundly needed (chapter **3**). The current and historical land covers (i.e., 1974 to 2010) were classified and SOC stocks under different land cover, soil types and slope position measured in the highland zone of south-west Rwanda. Under all soil groups, forest land uses (natural forest and plantation; 310 Mg C ha^{-1} and 372 Mg C ha^{-1}, respectively) held higher SOC stocks than agricultural land uses (annual crops, banana-coffee and tea; 127, 133 and 176 Mg C ha^{-1}, respectively). Forest clearing for agricultural use resulted in a loss of 272 (72 %) Mg C ha^{-1}. Slope position, explained partly the variation within agricultural land cover types. It was observed that SOC levels were best explained by contemporary land cover types, and not by soil group, conversion history or slope position, although the latter factor explained partly the variation within agricultural land cover types. Here marginal significant differences ($P<0.04$) in SOC stocks between the upper and lower slope positions were found. Lack of the influence of land use history on SOC stocks suggests that after conversion to a new land cover, SOC stocks reached a new equilibrium within the timestep that was observed (25 years). Changes in land use to agriculture will lead to changes in SOC stocks, generally tending towards a lower equilibrium status. With the conversion to annual crops, SOC concentration reached a new equilibrium at about 2.5 % SOC which is below the soil fertility threshold of 3 % that is considered the lower boundary in the region (Foster, 1971; Okalebo et al. 2002). This is implying that additions of SOC are required to increase the soil's effective fertility when they are subjected to growing annual crops.

6.4 Land use, soil erosion and sediment modeling

A new spatially explicit distributed hydrological model was created that couples a spatial daily soil water balance, the resulting groundwater and river flow, to event based erosion principles using a minimum dataset. By assuming that the rainfall and overland flow is active during a given number of hours per day, it was possible to calculate rainfall energy and runoff stream power, and use these to simulate sediment detachment, transport and deposition. There are two advantages to using rainfall energy and streampower principles: i), it enables a direct coupling with land use, which directly affects infiltration, splash protection and overland flow resistance, and ii) streampower principles allow the simulation of both detachment and deposition at pixel scale of 30x30 m while at the same time being able to route landscape sediment transport to catchment outlet and get a realistic soil loss estimate. In this way, we can avoid the simplified use of empirical factors in erosion modelling such as the C-factor (cover) and P-factor (management) of universal soil loss equation (USLE) type models or assumed sediment delivery ratios. The resolution of 30x30 m is in the order of agricultural field size (0.01 – 1.2 ha) in south west Rwanda which allowed the coupling of the model to the farm plot soil erosion and nutrient balance research, while at the same time calculate sediment and nutrient losses at catchment scale. The model performed reasonably well. Daily fluxes of peakflow, groundwater movement and baseflow were not always captured correctly but weekly average results much are better simulated. The prediction of when runoff actually occurs (on a weekly basis) is correct with a Nash-Sutcliffe efficiency of 68%. High levels of model accuracy could be achieved in the future with detailed data on the actual hydrology of the catchment. For example, in this case, rainfall data for modeling was simplified by measurements at three points in the catchment. A comparison of measured rainfall and peakflow data showed that at some points in time there was peakflow without recording the rainfall event that caused this flow. However, there is room for model and dataset improvements:

- a better spatial rainfall data coverage to capture convective rainstorms that cause erosion, which were only registered because peak flows were observed,
- a better representation of crop growth, either by crop growth modeling, or better field and satellite data; based on for instance NDVI time series.

Table 6.1. Total soil loss by percentage under land cover types in Rukarara catchment over the years (1974 - 2010)

Historical land cover type	Annual crops		Banana-coffee		Degraded Forest		Natural Forest		Plantation Forest		Savannas		Tea	
	tons	%	tons	%	tons	%	tons	%	tons	%	tons	%	tons	%
1974 land cover	282028	58	126854	26	0	0	57793	12	5344	1	5339	1	9982	2
1985 land cover	372455	68	82063	15	17779	3	49131	9	7054	1	5707	1	9943	2
1995 land cover	311880	64	74611	15	29669	6	34618	7	16342	3	3462	1	19503	4
2010 land cover	427061	77	28214	5	17246	3	45456	8	13713	2	1043	0	18498	3

Table 6.2. The effect of historical land cover type on total field and catchment scale soil

Historical land cover type	Total field scale soil loss (tons)	Mean soil loss at field scale (t/ha)	Soil loss at field scale compared to 2010 (%)	Total suspended sediment (TSS) load (tons)	Mean TSS load (t/ha)	Sediment Delivery ratio (SDR)	Soil loss at catchment scale compared to 2010 (%)
1974 land cover	487341	10	-12	142654	3	30	10
1985 land cover	544132	11	-1	134098	3	27	4
1995 land cover	490085	10	-11	121283	2	20	-6
2010 land cover	551232	11	0	129518	3	27	0

6.4.1 Soil loss and Total Suspended Sediment (TSS) loads between 1974 and 2010

Once the model was calibrated and verified, soil erosion rates were simulated over the period between 1974 and 2010. The period between 1974 and 2010 was chosen being the period for which high resolution land cover data was available for a detailed soil erosion studies. The 2010 catchment condition as the baseline and one year climate of 2010 to 2011 climate data was applied for simulation of the effect of land use type specific erosion rates (Table 6.1) and catchment scale soil loss (Figure 6.1a and b). Generally, soil loss between 1974 and 2010 was highest for annual crops, banana-coffee and tea crops, in declining order and lowest under forest and savannas land cover types. Most of the soil loss occurred on annual crops and banana-coffee land use types. Overall field soil loss had an increasing trend (over time) for annual crops, a declining trend for banana coffee and variable for other land use types (Table 6.1 and Figure 6.1a and b). There was no steady increase in total aggregated erosion, which depends entirely on the configuration of land use during the simulated period. Total aggregated soil loss was higher between 1985 and 2010 and was lowest in 1974 and 1995 (Table 6.2 and Figure 6.1a and b), largely influenced by land cover changes in the annual crops, banana-coffee, natural forest and plantation forest. Annual crops land cover increase during this period was offset by banana decrease, so these balance each other out, similarly natural forest declined but plantation forest increased (Figure 6.2). The changes for natural, plantation and degraded forest were rather dynamic but lower later than in 1974. Part of the plantation forest in 1995 was originally part of annual crop land between 1985 and 1995 but was cut-out as a buffer and planted with trees to protect the natural forest. At slope scale (Figure 6.3), most of the increase between 1974 to 2010 on annual crop and tea land cover occurred in the upper slope position. The total suspended sediment load (TSS) is slightly decreasing (Table 6.2). This trend is more related to the introduction of plantation forest between in 1985 and 2010 as a natural forest buffer around the forest and planting of trees in highly degraded farmlands where crop production was no longer possible (chapter 2) (Figure 6.2 and 6.3).

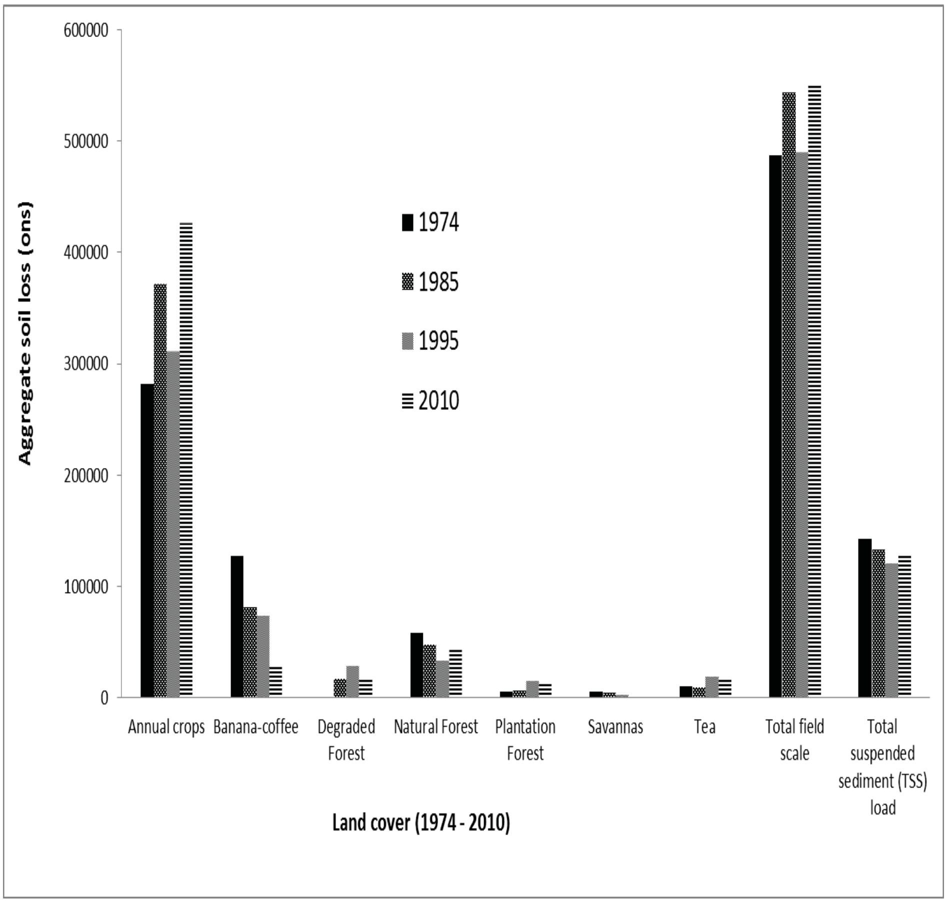

Figure 6.1a: Historical trends in soil loss by land cover type in Rukarara catchment (1974- 2010)

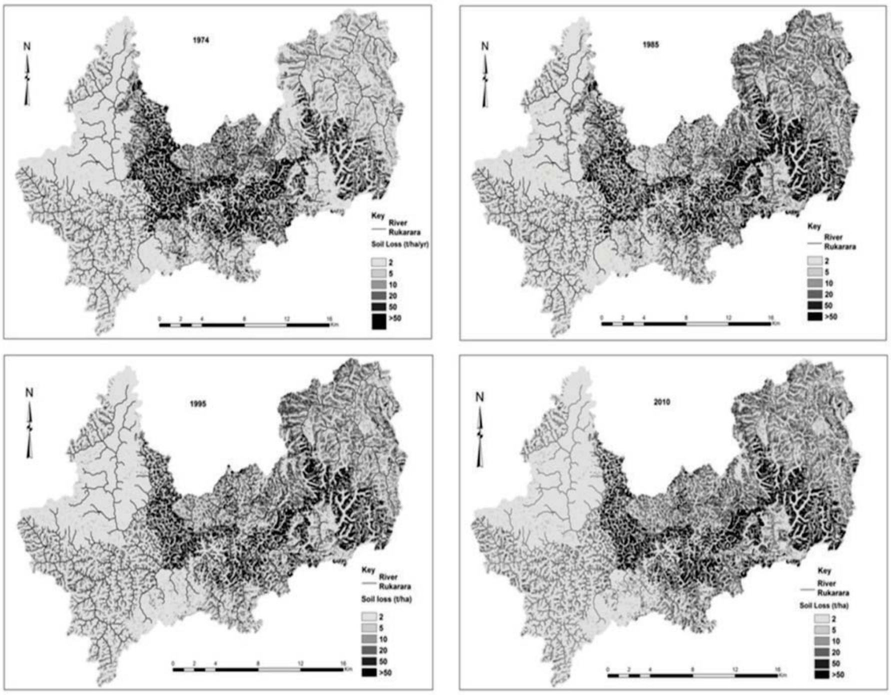

Figure 6.1b: Historical trends in soil loss by land cover type in Rukarara catchment (1974- 2010)

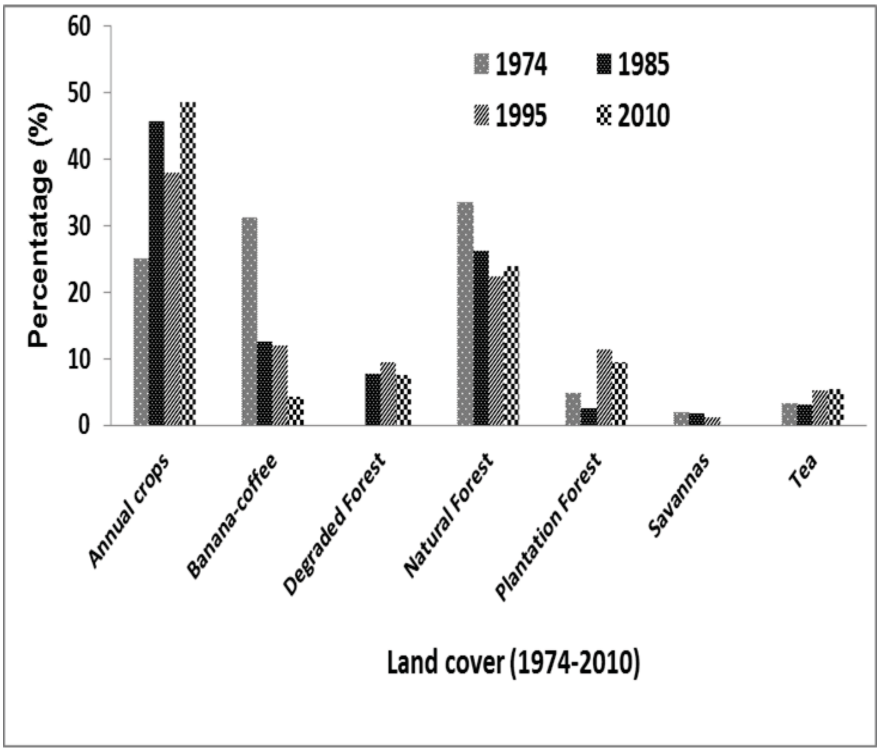

Figure 6.2: Percentages of land cover types and changes in Rukarara Catchment during 1974 - 2010

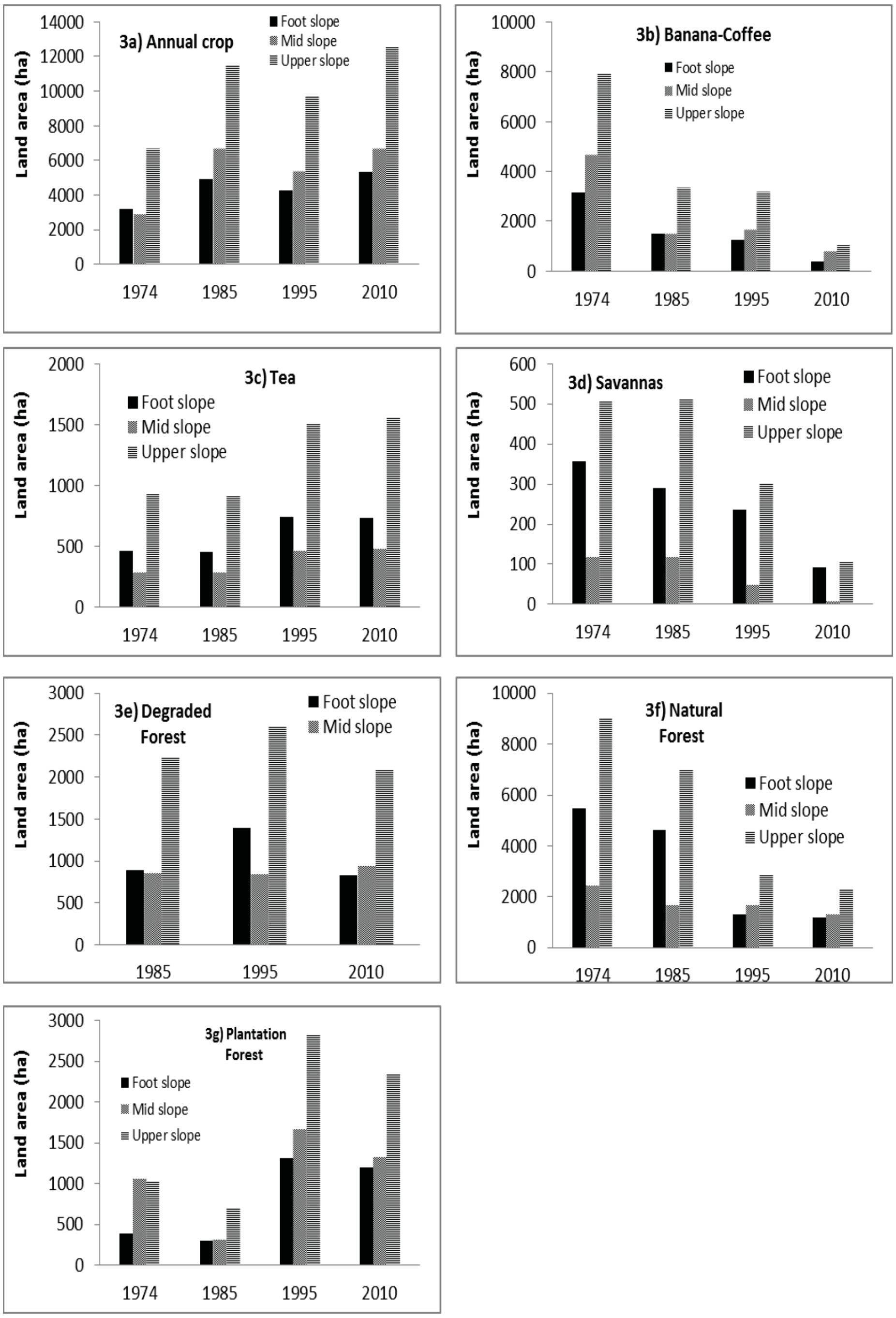

Figure 6.3: land cover changes at slope positions of Rukarara Catchment during 1974 - 2010

6.4.2 Spatial and temporal variability of soil loss in Rukarara catchment

Table 6.2 provides a comparison of soil loss at field scale and total suspended solid (TSS) load at the catchment scale. Aggregate field soil loss was higher than aggregate catchment scale soil loss. The TSS load in the year 2010 was lower than gross field erosion within the catchment, i.e., 27 % of the gross field is measured at the outlet. The 27% represents a sediment delivery ratio' (SDR). The 'Sediment delivery ratio' (SDR) defined as the TSS load at catchment scale divided by the gross erosion within the catchment. The difference between the level of gross erosion and TSS load leaving the catchment is a good indicator that not all the soil loss from fields reaches that catchment outlet. Only some fields deliver water and sediment directly to the stream. A factor that could explain the differences between soil loss within the catchment and suspended sediment load at the catchment outlet is the spatial organization of land cover in the catchment, because it may affect the connection between erosion zones and stream, allowing sediment interception and deposition within areas with high vegetation soil cover. The spatial patterns simulated period for 1974 to 2010 is presented in Table 6.3. Overall, the percentage of areas with tolerable soil loss (<10 t ha^{-1} yr^{-1}) declined from 78 % in 1974 to 74 % in 2010 indicating increasing soil degradation trend by soil erosion. The hotspot areas require intervention since they constituted up to 26 % of the catchment area in 2010. Further analysis of hotspot areas by slope position (Figure 6.4) reveals that soil loss was higher at the upper slope position followed by mid slope and lower slope. Slope position is a driving force in soil loss as can be seen from the spatial patterns. Other factors that could be the driving forces of soil loss are high rainfall intensities, soil properties, and poor agricultural practices. High amount of sediment was generated at the hilltops and upper slope segments as most agricultural land use types are situated in these areas and cover the largest area of the catchment (Figure 6.2 and 6.3). Annual crops on mid slope segments produced most soil loss expressed per hectare, but aggregated over the while catchment most sediment was generated at the hilltops and upper slope segments as most annual crops are situated in these areas (Figure 6.3 and 6.4).

Table 6.3: Spatial analysis of soil loss as influenced by land use changes during the period 1974 - 2010

soil loss classes (t/ha/yr)		1974	1985	1995	2010
Very Low (0 - 2)		45	39	42	39
Low (2 - 5)		21	20	21	20
Moderate (5 – 10)	12	15	14	15	
High (10 – 20)	9	12	10	11	
Severe (20 – 50)	7	8	7	8	
Very Severe (> 50)	6	6	6	7	
Tolerable soil loss of < 10 t/ha/yr (%)	78	74	77	74	
Hot spot of soil loss > 20 t/ha/yr	22	26	23	26	

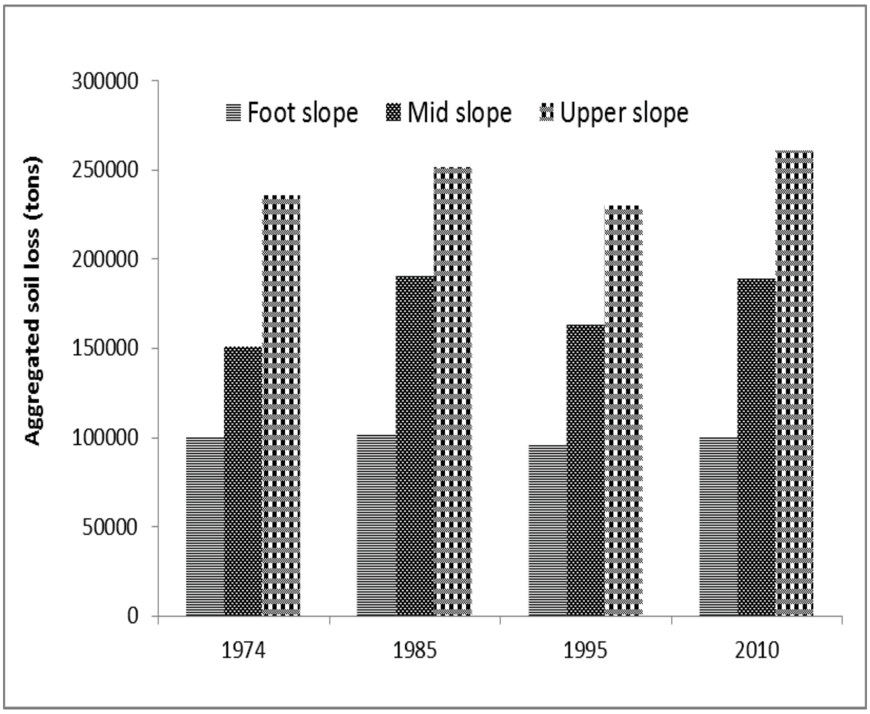

Figure 6.4: Soil loss by slope position during 1974 - 2010

6.5 The impact of land management practices on soil nutrient flows and balances

Intensification of agricultural practices combined with low fertilizer inputs is leading to an increased rate of soil degradation in south west of Rwanda (chapter **5**). Soils dominated by forest land use type have richer nutrient stocks. In places where land use is dominated by agricultural land use types, SOC in the top 20 cm soil depth lies between 23 to 40 gkg^{-1} i.e., nearly 30 - 50 % of the amount found in adjacent forest land use types (66 to 72 $g\ kg^{-1}$). Differences of similar magnitude were observed for stocks of soil nitrogen. Total soil nitrogen in agricultural land use types lies between 1.9 to 3.1 gkg^{-1} which is nearly 30% of the total nitrogen stocks found in adjacent forest land use types (9 gkg^{-1}). The low nutrient status in agricultural soils indicates on going nutrient depletion in agricultural land use systems. Assessment of Nitrogen (N), phosphorus (P) and Potassium (K) flows and balances for five different land use systems at both plot and farm level provide estimates at smaller scales. The wealth level of farm families and the slope position were used to further stratify the area. Nutrient mining by crops ranged from -82 to -30 kg N $ha^{-1}yr^{-1}$, -13 to +37 kg P $ha^{-1}yr^{-1}$, -56 to +34 kg K $ha^{-1}yr^{-1}$. The trend of N mining was highest for napier grass and lowest for legume – banana. A positive P and K balance was only observed in banana – coffee and tea that are intensively managed as commercial crops. Farm partial balances was marginally positive (8 kg N ha^{-1} yr^{-1}, 3 kg P ha^{-1} yr^{-1} and 8 kg K ha^{-1} yr^{-1}) and was low among food crops. N and K overall mean full balances were

Table 6.4. Simulated impact of land management practices on total field and catchment scale soil and nutrient losses

potential soil conservation practices	Total field scale soil loss (tons)	Mean soil loss at field scale (t/ha)	Soil loss at field scale compared to 2010 (%)	TSS (tons)	Mean TSS load (t/ha)	SDR	Soil loss at catchment scale compared to 2010 (%)	Soil nutrient losses at field scale (kg/ha)			Soil nutrient losses at catchment scale/ outlet (kg/ha)		
								N	P	K	N	P	K
2010 land cover	551232	11	0	129518	3	27	0	67	17	147	16	4	35
Sloping Terraces	512450	10	-7	84141	2	20	-35	62	16	84	10	3	22
Agroforestry at slope >35%	389760	8	-29	123291	2	25	-5	48	12	137	10	3	22
Mulching	339080	7	-38	87042	2	29	-33	41	11	90	15	4	32
Terracing and mulching	300707	6	-45	47225	1	17	-64	37	9	52	12	3	25
Agroforestry at hotspot area with soil loss > 20 $tha^{-1}yr^{-1}$	299258	6	-46	66219	1	17	-49	36	9	61	10	3	22
Agroforestry at slope >35% and mulching	256155	5	-54	83150	2	40	-36	31	8	80	8	2	18

generally negative across cropping systems while P was positive (– 36 kg N $ha^{-1}yr^{-1}$, +2 kg P $ha^{-1}yr^{-1}$, -8 kg K $ha^{-1}yr^{-1}$). The P and K mining occurred only in farm groups with food crop production and was positive for the intensively managed tea-annual cropping system farm group. In relation to the nutrient stocks, the results are less alarming for all nutrients with depletion rates of about 1%. Nutrients with depletion rate of more than 1% yr^{-1} are considered unsustainable for agroecosystem and therefore require intervention to restore the balance (Haileslassie et al., 2007). Among wealth groups, nutrient losses were highest for rich farmers compared to medium and poor farmers. Considering landscape positions, more nutrient loses occurred in the mid-slope compared to the upper slope or foot slope. Nutrient losses pathways were mainly the result of: erosion, denitrification and leaching. Erosion contributed the highest nutrient losses in the balance as follows; for P (7 kg $ha^{-1}yr^{-1}$), K (72 kg $ha^{-1}yr^{-1}$) and for N (23 kg $ha^{-1}yr^{-1}$). Historical nutrient balances could not be estimated due to lack of long-term crop yield and soil management data.

6.6 Implications of the current nutrient mining and soil erosion for sustainable land management

Land use and land cover changes in south west Rwanda drive SOC and nutrient depletion, and increased soil erosion on agricultural uses with low soil fertility. SOC stock in agricultural soil is much lower than there capacity when compared to adjacent forest soils. In relation to the nutrient stocks the results are less alarming for all nutrients with depletion rates of about 1% per year although this trend should not be allowed to continue to deplete nutrients. Erosion seemed to be less severe than anticipated although any loss of soil, however small, is likely to be serious to soil fertility management since soil erosion takes away the top soil that is richer in nutrients.

This study highlights the need to consider soil fertility restoration through integrated soil fertility management to improve nutrient capital on smallholder farms of south west Rwanda. SOC concentration is a proxy for soil fertility. The first focus of restoring soil fertility is to increase soil organic matter input so as to increase SOC content and consequently increase soil structure, nutrient holding capacity and soil biodiversity that is important for nutrient recycling. Examples of recommended management practices (RMP) that can enhance SOC stocks and improve soil quality can include (Lal, 2007; Smaling et al. 2012): conservation tillage, mulch farming, cover crops, integrated nutrient management that may include use of manure and compost, agroforestry and anti-erosion systems. This should be followed by improved nutrient inputs. Under the current soil fertility management practices, nutrient depletion rates of about 1% per year will continue over the next years, especially in annual crops. The management of N, P and K nutrient balance close to zero is to be preferred and stop nutrient depletion, but the road towards achieving that can be different. To improve nutrient input, smallholder farmers need actionable strategies. Soil mining has to be minimized through sustainable land management (SLM) techniques that will increase nutrient inputs, ensure nutrient cycling, reduce erosion and gradually increase soil organic matter content. Examples of commonly promoted and adopted SLM for soil and water conservation in Rwanda include: proper management of crop residues and integration of crop and livestock farming along with N fixing crops, contour ridges, sloping terraces, agroforestry, mulching, grass strips (Kagabo et al., 2012). Intercropping of particular tree species into annual food crop systems to sustain a green cover on the soil throughout the year, can bolsters soil erosion control and nutrient supply through nitrogen fixation and nutrient cycling (Roose and Ndayizigiye, 1997).

Policy initiatives to prevent land degradation in Rwanda date back to the early twentieth century, when planting trees and constructing trench ditches were already being promoted to prevent erosion (Bizoza, 2011). Sloping terracing, grass strips, agroforestry and straw mulching have been widely adopted and found effective at plot scale in increasing soil water storage and yield, while reducing erosion in the highlands of Rwanda (Roose and Ndayizigiye, 1997; Kagabo et al., 2012). Simulations were made to test the potential impact of commonly adopted soil and water conservation practices on soil and nutrient loss at catchment scale as follows; (1) sloping terraces on agricultural land use at slope angles (radians) of; 0.05, 0.10, 0.15 0.2,0.25, 0.3, 0.35, 0.40 and 0.45; (2) complete agroforestry in the annual and banana-coffee land cover type for whole areas of 2010 land use map with slope greater than 35%; (3) complete agroforestry in the annual and banana-coffee land cover type for areas of 2010 land use map with soil loss greater than 20 t $ha^{-1}yr^{-1}$ (hotspot areas); (4) mulching practice in annual crop and banana-coffee land cover types and (5) a combination of mulching practice and either of the soil conservation practices applied in annual and banana-coffee land cover type. Appropriate parameters were modified in the model input files such as management, crop database file, slope and land cover to implement potential future soil and water conservation practices for erosion control. Simulations outputs for alternative soil and water conservation scenarios were then compared against land cover of 2010 (baseline scenario) to compute percentage change in average values of soil erosion rates. Calculations were made to estimate nutrient losses resulting from soil loss. The results indicate that all soil conservation practices are capable of controlling soil erosion both at field and catchment scale (Table 6.4 and Figure 6.5, 6.6) and catchment scale (Table 6.4). The reduction in the soil nutrient loads was consistent with the reduction of sediment yield soil conservation practices implemented in all the scenarios (Table 6.4). Sloping terraces when implemented alone were less effective in reducing soil loss. This could be brought about by increased below terrace erosion which occurs when overland flow filtering through from the upper terraces causes more soil detachment and transport on the lower terrace. The cumulative effect of the overland flow from upper terraces is increased soil loss over the whole catchment. Various combinations of mulching and agroforestry or sloping terracing reduced aggregate field erosion by 7 % to 54 % of the values in 2010 and sediment load from 27 % to 17 % (SDR) (see Table 6.4 and Figure 6.5, 6.6). Nutrient losses were higher at field scale compared to catchment scale. The relationship between field erosion and river TSS was also related to N, P and K nutrient losses between these two scales. Nutrient loses by soil erosion occur when dissolved nutrients run-off and nutrient loaded sediment are exported out of agricultural area. During planning for soil conservation, lower SDR are preferred. From Table 6.4, the best simulated impact of soil conservation on SDR was observed for a combination of sloping terraces and mulching or agroforestry applied on catchment areas with soil loss >20 $tha^{-1}yr^{-1}$. With the implementation of soil conservation practices, the bulk of erosion consists of sediment redistribution within the catchment and sediment export to the river network is limited. Implementation of soil and water conservation practices on midslope and upper slope position that vulnerable to soil erosion, not only cut down erosion but also reduced the connectivity of flow from field to field. These soil conservation practices act as a barrier to run-off, increasing infiltration and decreasing flow volumes and speed, and ultimately reducing the transport capacity and encouraging sediment deposition. In the end these influence the rate of runoff, sediment and soil nutrient losses. These findings suggest that the implement soil and water conservation practices in the complex hillslopes of Rwanda and LVB could have an effect on preventing soil erosion and nutrient loses.

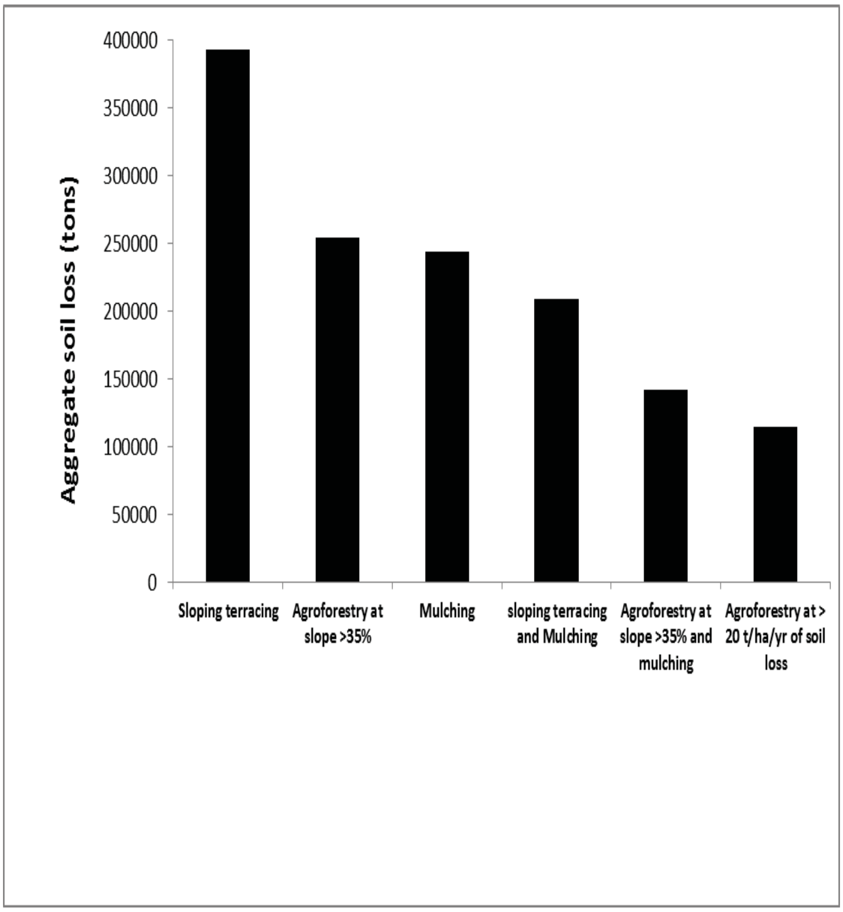

Figure 6.5: Potential impact of soil conservation practices on expected soil loss expected in annual crops land cover type in Rukarara catchment

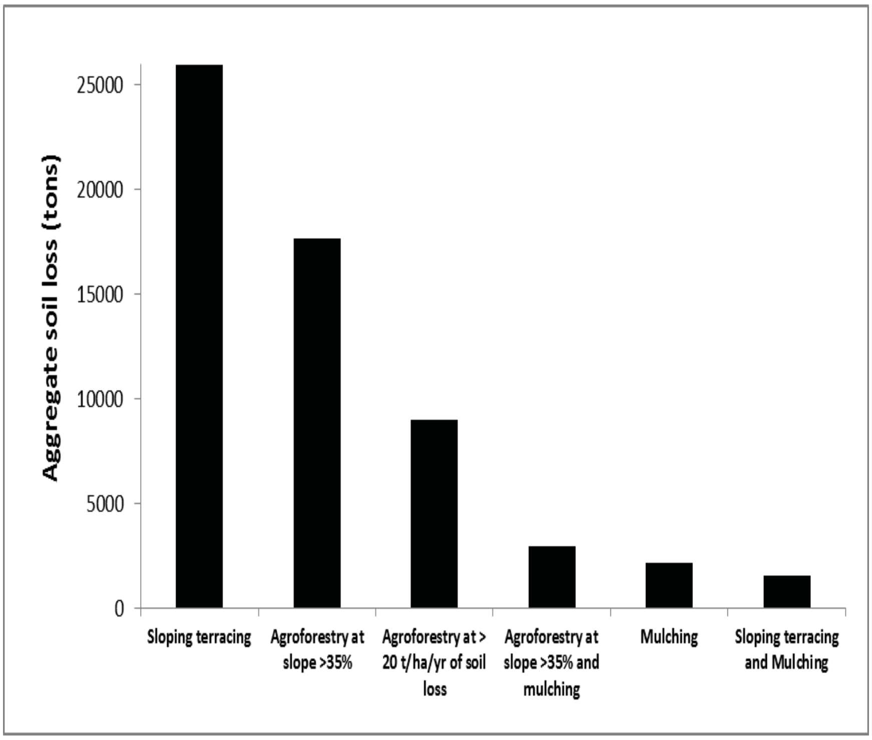

Figure 6.6: Potential impact of soil conservation practices on soil loss expected in Banana-Coffee land cover type in Rukarara catchment

6.7 Conclusion and recommendations

The trend of land use and land cover change that is mainly towards agricultural land use in south west Rwanda has been increasingly important in SOC and nutrient depletion, sediment and nutrient loading over the past years. The modelling of soil erosion in this Rukarara catchment points to the fact that there is no trend in increased soil erosion, TSS in decreasing and nutrient loading may therefore decrease. At basin scale, it can be speculated that increased in sediment load over the years into the lake is not self-evident. Converting forest land to agriculture does not necessarily trigger increased soil loss and sediment loading at the scale of Lake Victoria basin. For large catchment of Lake Victoria basin, the land cover configuration and varying terrain affects the connection between erosion zones and stream, allowing sediment interception and deposition within areas with high vegetation soil cover, flat, farm plots with soil conservation measures and along the stream channels. The river floodplains (such as Kagera river) leading to the main channel systems in larger river basins can represent an increasingly important sediment sink, which will attenuate changes in upstream sediment inputs. So, the larger the area, the greater the likelihood of sediment deposition along the way, which may even result in lower sediment yields at the basin outlet compared with the erosion rates measured on-site in erosion plots. Small catchments at the immediate periphery of the lake could be the most important source of sediment and pollution loading into the lake. Effort aimed at controlling sediment and pollution loading into Lake Victoria could pay more attention to small catchment at periphery.

The modelling presents only the physical effectiveness of soil and water conservation measures. The measures can be effective regarding reduction in runoff, sediment yield and soil nutrient losses in the catchment, but decisions to adopt would depend on socio-economic factor e.g., financial efficiency of the measures and resource availability with land owner. The model can be used as "policy support tool" to allow examination of the potential farm and catchment-scale impacts of varying farm land management practices, and of changing the land use mosaic on soil erosion control. The model suffers partly the same problems like other distributed models related to equifinality but has advantage of using a minimum input dataset and therefore the results should be interpreted in the light that;

- This is a daily continuous large scale model that requires a minimum dataset for erosion estimation which is an improvement,
- This is a spatial model which has big advantages, but this is subject to uncertainty and equifinality principle. It probably suffers from this because daily groundwater and runoff are not so well predicted.

Nutrient mining occurs in all cropping systems of south west highlands of Rwanda since soil nutrient stock has to offset the negative balances each year. Self-reinforcing interactions between soil erosion, fertility depletion, and woody vegetation cover decline preclude the possibility of sustainable agricultural production in this region. Without soil erosion control and fertilizer inputs (chapter **5**), soil fertility for increased crop yield will be difficult to imagine in south west hillslopes of Rwanda where soils under agriculture are inherently low in fertility and soil organic carbon (SOC) (Chapter **3**). Restoring soil fertility requires intensive nutrient and organic matter management and soil conservation to rebuild soil structure, to restore soil tilth and to raise soil fertility levels. Increased SOC concentration is a proxy for high soil fertility. The emphasis on protecting and improving soil fertility so it can produce crops and feed an ever-growing population often comes down to the level of SOC stocks. The magnitude of SOM stocks (and thus SOC stocks) in the soil is central to soil quality and productivity because it maintains to a large extent many soil physical properties (e.g. aeration, water retention, formation of stable aggregates), provides substrate to micro biota and plays a role in nutrient buffering and supply. Priority should be given to activities that provide incremental improvement of SOM and soil production capacity: e.g., NPK fertilizer additions, improved fallows, biomass transfer, and diversification of production. Promotion of anti-erosion systems (e.g., mulching, organic inputs, terraces, hedgerow barriers, agroforestry) commonly used by farmers in the region could reduce non-point source pollution loads and erosion of soil fertility.

Overall, the major contributions of this thesis were that it;

- Provided a spatially explicit historical assessment of LUCC in the last 100 years for Kagera basin region that is data scarce and is increasingly facing LUCC,
- Provided reliable estimates of smaller scale long-term losses of SOC and nutrient stocks from different land use systems among smallholder farms and forest land use type in south west Rwanda
- Developed and tested a new spatially explicit distributed hydrological model for erosion modelling
- Estimated in spatially explicit quantitative terms the large scale erosion rates and general patterns of soil loss in south west highlands of Rwanda.

Future research in this catchment should include identification of field effective soil and water conservation strategies specific to soil characteristics of different resiliencies. Assessment of socio-economic feasibility of the proposed soil and water strategies will help support decision making. So far the new spatially explicit hydrological model developed was calibrated on the basis of daily rainfall and run-off data. Immediate improvements can be done in terms of better spatial rainfall data coverage to capture convective rainstorms that cause erosion, but are now only registered seen as peak flow. A better representation of crop growth, either by crop growth modeling or better field and satellite data; for instance NDVI time series could improve model performance. Crop growth was generalized into a single change of crop cover in the growing seasons for all annual crops. More complex models need more data and are more affected by this. Furthermore, a targeted field validation of soil loss predicted hotspots could improve accuracy of model predictions.

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Summary

In the past four decades, the global community has been concerned over the state of the world's land resources to sustain the ever increasing global population and worsening world food situation. This is based on the available alarming figures of global estimates of land degradation and vulnerability to degradation processes. An enormous amount of empirical data is available on physical land degradation worldwide. Most is plot- or site-specific and quite difficult to aggregate and interpret on a national, regional or global basis. There are few systematic measurements of land degradation extent and severity of Sub-Saharan Africa (SSA) and South Asia (SA) where the problem is most severe. The Lake Victoria basin (LVB) is a prime example of an area experiencing high land and water degradation, increasing population pressure on land resources and as a result of that increasing food and energy demands. These are a threat to the Lake ecosystem productivity and livelihoods on the long term. Anecdotal evidence shows that this region experienced large-scale land clearing and conversion of native vegetation to agricultural land use over the last 50 years. Land use and land cover change (LUCC) is perceived to be a driver of land and water degradation. However, the consequences of LUCC on water and nutrient cycles are largely unstudied. Lack of consistent long-term LUCC data was always considered a critical gap in our knowledge of the nature and extent of soil degradation and soil erosion dynamics in the LVB. In this thesis, I interpreted combination of ancillary data and satellite imagery (Landsat) to reconstruct and analyze spatially explicit LUCC dynamics for the last century. The results showed that large trends of LUCC have rapidly occurred over the last 100 years. The most dominant LUCC processes were gains in farmland areas (60%) and a net reduction in dense forest (7% to 2.6%). The overall farmland expansion rate of 0.57% yr^{-1} is not as high as reported in other studies for SSA but at certain periods in the studied time frame, higher rates were observed (1.56 % yr^{-1} between 1974 and 1985). The results of this study offer important directions for follow up research aimed at quantifying impacts of LUCC on depletion of soil organic carbon (SOC) and nutrient stocks, and increase in sediment and nutrient loading in the river system. In subsequent studies, I targeted these issues. How changes in LUCC impact on soil C stocks in the Lake Victoria landscapes was originally unclear. In the south west highlands of Rwanda, forest clearing for agricultural use resulted in a loss of 272 (72 %) Mg C ha^{-1}. It was observed that SOC levels were best explained by contemporary land cover types, and not by soil group, conversion history or slope position, although the latter factors explained partly the variation within agricultural land cover types. Lack of the influence of land use history on SOC stocks suggests that after conversion to a new land cover, SOC stocks reached a new equilibrium within the timestep that was observed (25 years). SOC stock declined under transitions from banana-

coffee to annual crop by 5 % (6 Mg C ha^{-1}) or under transitions from natural forest to degraded forest by 21 % (66 Mg C ha^{-1}) and increased for transitions from annual crops to plantation forest by 193 % (245 Mg C ha^{-1}). Next to lack of information on SOC stocks, catchment scale and long-term efforts to estimate erosion rates are missing and general patterns of soil loss unstudied. This was coupled with the problem that the current spatially distributed erosion models are often not (yet) capable of continuous simulation of long periods because they do not always include a daily water balance, plant growth and development. This leads to difficulties in incorporating a high degree of spatial information, especially land use information, so that the effects of land use changes on soil erosion have hitherto not been investigated in detail using models. Therefore, a new spatially explicit distributed hydrological model was created that couples a spatial daily soil water balance to event based erosion principles using a minimum dataset. By assuming that the rainfall and overland flow is active during a given number of hours per day, it is possible to calculate rainfall energy and runoff stream power, and use these to simulate sediment detachment, transport and deposition. The model can simulate terrestrial hydrology; infiltration, runoff processes and erosion patterns as daily outputs. The model provides output information about where soil moves to and about deposition. The model predictive capability of reproducing field situation of peakflow, discharge and sediment delivery by surface runoff from hillslopes to river is reasonable with a Nash-Sutcliffe efficiency of 68%. The simulated field level sediment dynamics are in line with soil loss found in many field studies in similar environments in Rwanda and the LVB with a sediment delivery ratio of 27%. The pattern of soil loss by slope position was important under annual crop land use types and soil loss was highest in the midslope, which concurs with the effect of agricultural land uses found on SOC stocks. The model output was used for a detailed spatial assessment of sedimentation and soil loss feedback at 30 x 30 m pixels as input in the nutrient balance model NUTMON (nutrient monitoring). The resolution of 30x30 m is in the order of agricultural field size (0.01 – 1.2 ha) in the Lake Victoria basin which allowed the coupling of the model to the farm plot soil erosion and nutrient balance research, while at the same time calculate sediment and nutrient losses at catchment scale. The NUTMON model assesses the net nutrient balance of five inflows (mineral fertilizer, organic inputs, atmospheric deposition, nitrogen fixation and sedimentation) and five outflows (crop products, crop residues, leaching, gaseous losses and erosion). Nutrient balances were assessed for 91 farm households which together covered 506 small scale plots (ranging from 0.01 to 1.2 ha) distributed across five cropping systems in south-west Rwanda. Nutrient mining by crops ranged from -82 to -30 kg N $ha^{-1}yr^{-1}$, -13 to +37 kg P $ha^{-1}yr^{-1}$, -56 to +34 kg K $ha^{-1}yr^{-1}$. The trend of N mining was highest for napier grass > tea > banana-cereal > fallow > cereals > root – tuber > banana –

coffee > banana – bean > legume – banana. N and K overall mean farm full balances were generally negative across cropping systems while P was positive (– 36 kg N $ha^{-1}yr^{-1}$, +2 kg P $ha^{-1}yr^{-1}$, -8 kg K $ha^{-1}yr^{-1}$). Erosion contributed the highest nutrient losses in the balance as follows; for P (7 kg $ha^{-1}yr^{-1}$), K (72 kg $ha^{-1}yr^{-1}$) and for N (23 kg $ha^{-1}yr^{-1}$). The self-reinforcing interactions between LUCC, soil erosion, fertility depletion, and woody vegetation cover decline preclude the possibility of sustainable agricultural production in south-west Rwanda and across the LVB. Without soil erosion control and fertilizer inputs, agricultural development will be difficult to imagine in the region where soils under agriculture are inherently low in fertility and SOC stocks. In places where land use is dominated by agricultural land use types, SOC in the top 20 cm soil depth lies between 23 to 40 gkg^{-1} i.e., nearly 30 - 50 % of the amount found in adjacent forest land use types (66 to 72 g kg^{-1}). Differences of similar magnitude were observed for stocks of soil nitrogen. Total soil nitrogen in agricultural land use types lies between 1.9 to 3.1 gkg^{-1} which is nearly 30% of the total nitrogen stocks found in adjacent forest land use types (9 gkg^{-1}). Restoring soil fertility requires intensive nutrient and organic matter management and soil conservation to rebuild soil structure, to restore soil tilth and to raise soil fertility levels. Promotion of anti-erosion systems commonly used by farmers in the region could reduce non-point source pollution loads and erosion of soil fertility, and gradually increase soil organic matter content by retaining sediments. Simulations were made to test the potential impact of commonly adopted sustainable land management (SLM) techniques for soil and water retention in Rwanda such as sloping terraces, agroforestry and mulching or in combination for erosion control. Results indicated that a combination of mulching and agroforestry or terracing reduces erosion by 7% to 64 % of the values in 2010 (base year) and sediment delivery ratio from 27 % to 17 % sediment delivery ratio. The impact of SLM techniques on erosion control was also directly related to reduction of N, P and K nutrient losses. Overall, the major contributions of this thesis were that it;

- Provided a spatially explicit historical assessment of LUCC in the last 100 years for Kagera basin region, upper catchment of LVB that is data scarce and is increasingly facing LUCC,
- Provided reliable estimates of smaller scale long-term losses of SOC and nutrient stocks from different land use systems among smallholder farms and forest land use types in south west Rwanda
- Developed and tested a new spatially explicit distributed hydrological model for erosion modeling
- Estimated in spatially explicit quantitative terms the large scale erosion rates and general patterns of soil loss in south west highlands of Rwanda.

Samenvatting

De afgelopen vier decennia is er internationaal gezien bezorgdheid geweest over de capaciteit van de aarde om immer groeiende wereldbevolking te voorzien in zijn behoeften en de verslechterende voedsel situatie. Deze bezorgdheid komt voort uit globale schattingen van land degradatie en kwetsbaarheid voor degradatie processen. Een grote hoeveelheid empirische gegevens is wereldwijd beschikbaar over fysieke land degradatie. De meeste gegevens zijn specifiek voor een klein gebied en lastig te interpreteren op nationaal, regionaal of globaal niveau. Er zijn slechts weinig systematische kwantificeringen van land degradatie bezuiden de Sahara en Zuidelijk Azië, waar het probleem het grootst is. Het Victoriameer Bassin (LVB) is een goed voorbeeld van een gebied waar op grote schaal land- en water degradatie, een groei in bevolking en een groeiende vraag naar voedsel en energie plaats vind. Deze ontwikkelingen vormen een bedreiging voor de productiviteit van het omliggende ecosysteem en het levensonderhoud van zijn bewoners op de langere termijn. Anekdotische aanwijzingen suggereren dat de afgelopen vijftig jaar grootschalige omzettingen van natuurlijke vegetatie naar landbouw hebben plaatsgevonden in dit gebied. Landgebruik en landgebruiksveranderingen (LUCC) worden gezien als een hoofdoorzaak voor land- en water degradatie. De gevolgen van LUCC voor de nutriënten- en waterhuishouding zijn echter nauwelijks onderzocht. Gebrek aan systematische lange termijn gegevens over LUCC werden altijd gezien als een belangrijke omissie in de huidige kennis over bodemdegradatie en bodemerosie dynamiek in het LVB.

In dit proefschrift, heb ik een combinatie van satelliet beelden (LandSat) en aanvullende bronnen gebruikt om een ruimtelijk expliciete reconstructie van landgebruiksveranderingen in het LVB van de afgelopen eeuw te maken en te analyseren. De resultaten lieten zien dat grote landgebruiksveranderingen in korte periodes plaats vonden in de afgelopen eeuw. De belangrijkste landgebruiksveranderingen waren toenames in landbouw areaal (60%) en een netto reductie in primair bos (7% naar 2.6%). De gemiddelde aanwas van landbouwareaal over de hele periode (0.57% jaar^{-1}) is niet zo hoog als gerapporteerd voor andere gebieden bezuiden de Sahara, maar tijdens sommige periodes werden veel hogere omzettingssnelheden waargenomen (1.56% jaar^{-1} tussen 1974 en 1985). Deze resultaten bieden belangrijke inzichten voor vervolgonderzoek gericht op het kwantificeren van de gevolgen van landgebruiksveranderingen op bodemorganische stof (SOC) en nutriënten huishouding, en de toename in sediment en nutriënten lading in het riviersysteem.

In vervolg studies heb ik naar deze aspecten gekeken. Hoe landgebruiksveranderingen bodem organische stof beïnvloedden was

voordien onduidelijk. Ontbossing ten behoeve van landbouw zorgde voor een verlies van 272 (72%) Mg C ha^{-1} in de zuidwestelijke hooglanden van Rwanda. We vonden dat bodemorganische stof het beste verklaard kon worden door huidig land gebruik, en dat voormalig landgebruik, bodem type of positie op de helling geen effect hadden, hoewel de laatste wel een klein effect had als alleen naar landbouwgronden werd gekeken. Het feit dat voormalig landgebruik geen effect lijkt te hebben op de huidige hoeveelheden bodemorganische stof suggereert dat er na omzetting van het ene naar het andere landgebruikstype snel (binnen een tijdstap van 25 jaar) een nieuw (lager) evenwicht wordt bereikt. De hoeveelheid bodemorganische stof werd minder onder de omzetting van banaan- en koffieplantages naar eenjarige gewassen (-5%, 6Mg C ha^{-1}) en onder de omzetting van primair bos naar secundair bos (-21%, 66 Mg C ha^{-1}) maar nam toe bij de omzet van eenjarige gewassen naar bos plantages (+193%, 245 Mg C ha^{-1}).

Naast het gebrek aan informatie over bodemkoolstof, zijn er nauwelijks pogingen ondernomen om erosie op de schaal van een stroomgebied in kaart te brengen. Daarnaast bestond het probleem dat de huidig beschikbare ruimtelijk expliciete erosiemodellen vaak (nog) niet in staat zijn om lange periodes na te bootsen omdat ze geen dagelijkse waterbalans bevatten, of planten groei meenemen. Hierdoor is het lastig om gedetailleerde ruimtelijke informatie, voornamelijk landgebruiksinformatie, in de simulaties mee te nemen, waardoor de effecten van landgebruiksveranderingen op erosie nog weinig met modelstudies zijn onderzocht. Daarom is er een nieuw, ruimtelijk expliciet hydrologisch model ontwikkeld dat een ruimtelijke dagelijkse water balans koppelt aan erosiegebeurtenissen en waarvoor een minimale dataset nodig is. Door de aanname te maken dat neerslag en oppervlakkige afstroming actief zijn gedurende een bepaald aantal uren per dag, is het mogelijk om neerslag energie en afstroom kracht te berekenen. Deze kunnen gebruikt worden om sediment productie, transport en depositie te berekenen. Het model simuleert de terrestrische hydrologie en heeft infiltratie, afstroming en erosiepatronen als uitkomst. Dit geeft informatie over waar grond wordt verwijderd en waar het terecht komt. Het model kon veldwaarnemingen zoals piekafvoer, totale afvoer, sediment aflevering en oppervlakte afstroming naar de rivier met een Nash-Sutcliffe effectiviteit van 68% redelijk reproduceren. De nagebootste veranderingen in sediment op de schaal van een individueel perceel waren vergelijkbaar met veldstudies in vergelijkbare gebieden in Rwanda en het Victoriameer Bassin met een sediment afvoer van 27%. Onder eenjarige gewassen was de positie in het landschap van belang, waarbij de hoogte bodemverliezen optraden midden op de helling, een patroon dat overeenkomt met de bevindingen voor bodemorganische stof.

De modeluitkomsten gaven een gedetailleerde (30 m bij 30 m ruimtelijke resolutie) ruimtelijke schatting van bodem verlies en sedimentatie welke zijn gebruikt voor het maken van een schatting van de nutriënten balans met de NUTrienten MONitoring (NUTMON) methode. De gebruikte resolutie van 30m bij 30m komt overeen met de werkelijke perceel grootte die gevonden wordt in het Victoriameer bassin (0.01 -1.2 ha) wat het mogelijk maakte om tegelijkertijd zowel op huishoudniveau naar nutriënten balansen te kijken als op de schaal van het stroomgebied nutriëntverliezen te berekenen. Het NUTMON model schat de nutriëntenbalans op basis van vijf stromen naar het systeem toe (kunstmest, organische mest, atmosferische depositie, stikstof fixatie en sedimentatie) en vijf stromen het systeem uit (afvoer van gewassen, afvoer van gewasresten, uitwassing naar diepere bodemlagen, verliezen via de atmosfeer en erosie).

Er werden voor 91 boeren huishoudens nutriëntenbalansen opgesteld, welke in totaal 506 kleine percelen (tussen de 0.01 en 1.2 ha) bestreken, verspreid over 5 gewassystemen in het zuidwesten van Rwanda. Nutriëntverliezen op perceel niveau via gewassen varieerden tussen -82 tot -30 kg N ha^{-1} $jaar^{-1}$, -13 tot +37 kg P ha^{-1} $jaar^{-1}$, en -56 tot +34 kg K ha^{-1} $jaar^{-1}$. Stikstof (N) verliezen waren het hoogste voor Napier gras, gevolgd door (in aflopende volgorde)thee, banaan-granen, braak, granen, wortel- & knolgewassen, banaan-koffie, banaan-bonen en als laagste leguminosen-banaan. Gemiddeld genomen waren Stikstof en Kalium balansen vaak negatief op het niveau van een huishouden, terwijl Fosfor doorgaans positief was (- 36 kg N ha^{-1} $jaar^{-1}$, -8 kg K ha^{-1} $jaar^{-1}$ en +2 kg P ha^{-1} $jaar^{-1}$). Erosie droeg het meeste bij aan het verlies van nutriënten (23 kg N ha^{-1} $jaar^{-1}$, 72 kg K ha^{-1} $jaar^{-1}$ en 7 kg P ha^{-1} $jaar^{-1}$).

De versterkende koppelingen tussen landgebruiksverandering, bodemerosie, bodemvruchtbaarheid afname en ontbossing vormen een barrière voor duurzame landbouwproductie in zuidwest Rwanda en binnen het Victoriameer bassin. Zonder erosiepreventie en gebruik van kunstmest, in een gebied dat al een inherent lage bodemvruchtbaarheid kent, is het moeilijk om landbouwontwikkeling voor te stellen. Op plekken waar landbouw domineert is het bodemorganische stof gehalte van de eerste 20 cm tussen de 23 tot 40 g kg^{-1}, ongeveer 30 tot 50% van wat er in nabijgelegen onaangetaste bossen wordt gevonden (66 tot 72 g kg^{-1}). Vergelijkbare verschillen werden er gevonden voor bodemstikstof. De totale hoeveelheid stikstof in landbouwgronden ligt tussen de 1.9 tot 3.1 g kg^{-1}, wat ongeveer 30% van de stikstofhoeveelheden in nabijgelegen bossen is (9 g kg^{-1}). Het herstel van de bodemvruchtbaarheid vergt intensief beheer van nutriënten en bodemorganische stof, naast bodemconserveringsmaatregeling om de bodemstructuur weer op te bouwen. Promotie van in de regio gebruikelijke anti-erosie systemen zouden diffuse vervuiling van afstroomwater en erosie kunnen verminderen en, door het vasthouden van sedimenten, de bodemorganische stofgehaltes weer kunnen laten groeien.

Met het eerder genoemde erosiemodel zijn de effecten van veelvoorkomende anti-erosiemaatregelen zoals de aanleg van terrassen, gemengde land- & bosbouwsystemen, het bedekken van de bodem met gewasrestanten en de combinatie van deze maatregelen gesimuleerd. Het resultaat suggereerde dat een combinatie van bodembedekking en gemengde land- & bosbouw of de aanleg van terrassen de erosie terug kon brengen naar 7% tot 64% van de erosie in 2010 en de sediment afvoer van 27% naar 17%. Deze gevolgen werden ook gerelateerd aan een vermindering in verliezen van stikstof, fosfor en kalium.

Al met aal zijn dit de hoofdpunten uit dit onderzoek:

- Het gaf een ruimtelijke inschatting van de landgebruiksveranderingen de afgelopen 100 jaar in het Kagera stroomgebied, het bovenste stroomgebied van het Victoriameer bassin, waar gegevens schaars zijn en toenemende landgebruiksveranderingen plaatsvinden.
- Het gaf betrouwbare schattingen van lange termijn verliezen van bodemorganische stof en nutriënten voor verschillende landgebruikssystemen in zuidwest Rwanda.
- Er is een nieuw ruimtelijk explicit hydrologisch model ontwikkeld en getest om erosie te simuleren.
- Het geeft een ruimtelijk expliciete kwantitatieve schatting van de erosie snelheden en patronen op grote schaal in de hooglanden van zuidwest Rwanda.

Biography

John Ejiet Wasige was born on January 21st 1972 in Tororo, Uganda. After high school, he joined Makerere University for a BSc. Agriculture Degree between 1994 – 1998. During 1999 and part of 2000, he worked as an agricultural officer in the Ministry of Agriculture, Animal industry and Fisheries (MAAIF), Uganda. In October 2000, he was awarded a Rockefeller scholarship to pursue a MSc. in Soil Sciences at Makerere University for which he graduated in January 2004. After his graduating, he worked as a research associate on various projects and also taught at Makerere University. He has attended various international short courses in Earth Surface Modelling and Data Assimilation in biogeochemical, crop-weather and hydro-meteorological tools (e.g., Spatial-DSSAT, arcSWAT, LAPSUS, Spatial-CENTURY, NUTMON) for science-based monitoring of the environment. In 2006, he received a NFP scholarship to attend a short course on Principles of Spatial Data Handling: Databases, GIS and Remote Sensing at the International Institute for Geo-Information Science and Earth Observation (ITC), The Netherlands. It was during this time that he met Prof. E.M.A. Smaling who had just given a lecture on sustainable land management (SLM), an aspect that has been his area of professional interest. After discussion, Prof. E.M.A. Smaling accepted to supervise this PhD programme which resulted in this thesis. His PhD programme was jointly funded by The Netherland Fellowship Programme (NFP) and Makerere University.

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