KWAME NKRUMAH UNIVERSITY OF SCIENCE AND TECHNOLOGY

KUMASI, GHANA

Landscape Patterns and Modelling of Soil-Vegetation Relationship and Related Ecosystem Services to Support Landscape Conservation in the Mo River Basin (Togo)

by

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DECLARATION

I hereby declare that this submission is my own work towards the PhD degree in Climate Change and Land Use, and that, to the best of my knowledge, it contains no material previously published by another person, nor material which has been accepted for the award of any other degree of the University, except where due acknowledgment has been made in the text.

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ABSTRACT

In order to support integrated landscapes and restoration efforts, this research focused on the assessment and monitoring of the spatio-temporal land use/cover change (LUCC) and degradation in the Mo River Basin (a subunit of the Volta basin of about 1,490 km^2 in Central Togo). Field measurements, legacy and ancillary data were subjected to sequential multivariate methods, correlation analyses, geostatistics and modelling to analyse landscape conditions. First, along a gradient of land protection regime, data from extensive soil sampling and forest inventory were used to analyse soil organic carbon (SOC) and total nitrogen (TN) storage up to 30 cm depth, and the interactions between vegetation-soil conditions. Next, Landsat images of 1972, 1987, 2000 and 2014 combined with most updated global topographic and soil databases were used to analyse the landscape changes and its impacts on SOC, TN, soil loss potential and landscape patterns. Finally, the Landscape Management and Planning Tool adapted for the Mo basin (LAMPT_Mo), a spatially explicit model based on the Revised Universal Soil Loss Equation (RUSLE), was used to model the historical soil loss, and evaluate the efficiency of some land management scenarios. Different databases and field characterisation were used for model calibration and validation. The results showed that SOC and TN varied significantly according to land use/cover types, soil depths, topographical positions and land protection regime. With forests and woodlands exhibiting highest amounts of nutrients, mean TN varied from 0.06 to 0.16 % in the topsoil (0 – 10 cm) and 0.04 to 0.09 % in the subsoil (10 – 30 cm). Similarly, SOC ranged from 1.81 % in farmlands to 3.58 % in forests in the topsoil while woodlands had highest SOC in the subsoil (2.23 %). The river basin is made up of four and three vegetation types in unprotected and protected areas, respectively. The synergized effects of land protection status, soil conditions, landform, and human disturbances drive these vegetation patterns. From the historical analyses, natural lands dominated the basin, though their area constantly decreased since 1972. Contemporary LUC (in 2014) is dominated by savannahs/shrubs (53 %), woodlands (27 %) and forests (11 %). Non-cultivated and vegetation regrowth areas were the most dominant of the LUCC trajectories to whom SOC, TN and soil loss potential were responsive. Trajectories of land cover decline induced soil quality deterioration while correlation analyses showed soil loss to be more landform-driven than LUCC. Simulations using LAMPT_Mo yielded values of net soil loss (NSL) far higher than the tolerable limits for the Tropics. NSL markedly changed over time with about 26, 23, 27 and 44 Mg $ha^{-1}y^{-1}$, for 1972, 1987, 2000 and 2014, respectively. Steep slopes ($\geq 15^{o}$), poorly covered lands, and riversides (distances ≤ 100 m) are critical areas of sediment source. Some intervention measures such as controlling erosion hotspots through LUC protective measures could help reducing NSL up to 70 %, to the tolerable limits for the Tropics. The combination of methods and approaches used for the monitoring and assessment of landscape change and degradation enabled to capture the different spatial aspects of the problem of land degradation in the Mo basin. The study demonstrated that important appropriate conservation measures would be necessary for the catchment rehabilitation, protection and sustainable resource use.

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LIST OF ABBREVIATIONS

ADB = African Development Bank

ADF = African Development Fund

CCA = Canonical Correspondence Analysis

CCLU = Climate Change and Land Use

CGIAR = Consultative Group on International Agricultural Research

DBH = Diameter at Breast Height

DCA = Detrended Correspondence Analysis

DEM = Digital Elevation Model

DFD = Deforestation and Forest Degradation

DGSCN = Direction Générale de la Comptabilité et Statistique Nationale

DGE = Direction Générale de l'Energie

DRHE = Direction Régionale de l'Hydraulique et de l'Eau

ESS = EcoSystem Services

FAO = United Nations Food and Agriculture Organisation

GDP = Gross Domestic Product

GIS = Geographical Information System

GLASOD = Global Assessment of Soil Degradation

GLP = Global Land Project

GOFC-GOLD = Global Observation of Forest and Land Cover Dynamics

GSL = Gross Soil Loss

HWSD = Harmonised World Soil Database

IGN = Institut Geographique National

IIASA = International Institute for Applied Systems Analysis

ISRIC = International Soil Research and Information Centre

ISS-CAS = Institute of Soil Science – Chinese Academics of Sciences

ITTO = International Tropical Timber Organisation

JRC = Joint Research Centre

KNUST = Kwame Nkrumah University of Science and Technology

LADA = LAnd Degradation Assessment

LAMPT_Mo = LAndscape Management and Planning Tool adapted for the Mo basin

LAPMAT = LAndscape Planning and MAnagement Tool

LD = Land Degradation

LUC = Land Use/Cover

LUCC = Land Use/Cover Change

MEA = Millennium Ecosystem Assessment

MERF = Ministère de l'Environnement et des Ressources Forestières

MMF = Morgan-Morgan-Finney

MODIS = Moderate Resolution Imaging Spectroradiometer

NDVI = Normalised Difference Vegetation Index

NSL = Net Soil Loss

NTFP = Non-Timber Forest Products

OM = Organic Matter

PA/UPA = UnProtected Area/ Protected Area

RUSLE = Revised Universal Soil Loss Equation

SAGA = System for Automated Geoscientific Analyses

SDR = Sediment Delivery Ratio

SECE = Similar Environmental Constraints Envelop

SEM = Spatially Explicit Model

SOC = Soil Organic Carbon

SPOT = Système Pour l'Observation de la Terre

SRTM = Shuttle Radar Topography Mission

SSA = Sub-Saharan Africa

STCI = Sediment Transport Capacity Index

SWAT = Soil and Water Assessment Tool

TN = Total Nitrogen

TWI = Topographic Wetness Index

TWINSPAN = Two Way Indicator Species Analysis

UNCCD = United Nations Convention to Combat Desertification

UNFCCC = United Nations Framework Convention on Climate Change

USLE = Universal Soil Loss Equation

USPED = Unit Stream Power-based Erosion Deposition

UTM WGS = Universal Transverse Mercator World Geodetic System

WASCAL = West African Science Service Centre on Climate Change and Adapted Land Use

WEPP = Water Erosion Prediction Project

WOCAT = World Overview for Conservation Approaches and Technologies

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CHAPTER 1: GENERAL INTRODUCTION

1.1. Background

Worldwide as ever before, humans have increasingly altered land resources at various scales, through multiple land uses to meet increasing needs (Ellis et al., 2010; Foley et al., 2005). This transformation of the Earth surface has been so marked, engendering debates for the definition of a new geological epoch, the "*Anthropocene*" (Ellis et al., 2013; Gaia, 2011). In some cases, this transformation led to degradation i.e. a disruption between land capacity to support human pressures and its potential to continuously provide services. According to the United Nations Convention to Combat Desertification, land degradation (LD) is a *"reduction or loss of the biological productivity, resulting from land uses, or combination of processes, such as soil erosion, deterioration of properties of soil and long-term loss of natural vegetation*". With this definition, LD is regarded as a multifaceted phenomenon of major environmental concern. Its consequences are numerous, ranging from the economic aspects of soil quality and productivity loss (Costanza et al., 1997; Lal, 1997; Vlek et al., 2008) to the ecological aspects of the disruption in the biogeochemical cycles as well as landscape integrity (Lambin et al., 2003; Verburg et al., 2013).

LD, its factors and potential impacts have been investigated through various approaches, scales and dimensions (Gounaridis et al., 2014; He et al., 2014; Kim et al., 2013; Kim et al., 2014; Zheng et al., 2014; Zhou et al., 2014b; Zhou et al., 2012). In Sub-Saharan Africa (SSA), though little reliable data exist on the magnitude of LD associated with climate change and implications for food security (Connolly-Boutin & Smit, 2015), estimates indicated that it has affected almost 70 % of the productive lands with a constant increasing rate (Vlek et al., 2008). As good indicators, soil erosion (Le et al., 2012b; Tamene et al., 2014), deforestation and forest degradation (DFD)

(Chagumaira et al., 2015; Damnyag et al., 2013) and soil ecosystem services (ESS) loss (Abdel Kawy & Ali, 2012) were used to assess LD severity and extent, induced by poor land allocation and use, inappropriate land policy and planning strategies (Petursson et al., 2013; Romero-Ruiz et al., 2012; Vedeld et al., 2012).

Despite many attempts, tackling LD is still elusive and unmanageable, especially in the context of climate change. Profound changes still affect the spatial patterns and functions of ecosystems requiring continuous assessment and evolvement of new methodological considerations (Verburg et al., 2013; von Stechow et al., 2015). As ecosystems are increasingly evolving new processes in response to disturbances (Masera et al., 2015), there is a growing need for permanent and continuous monitoring of land ecosystems in order to improve land resilience and mitigate LD through interdisciplinary and integrated approaches (Braimoh & Osaki, 2010) that take into account spatially explicit and multiscale interactions (Turner et al., 2015). Thus, combination of historical multi-source data with contemporary field records and innovative management and planning strategies could offer new paradigm and holistic approach for conservation and restoration (Cellier et al., 2011; Tamene et al., 2014).

1.2. Problem statement

1.2.1. National and local challenges of land degradation

In Togo, LD is commonly observed through DFD and soil degradation in terms of erosion by water and nutrient depletion. Most of the recent studies showed that DFD affects all landscapes including protected areas (Adjonou et al., 2010; Badjana et al., 2014; Folega et al., 2015; Folega et al., 2014b). For the period 2005-2010, Togo had an alarming deforestation rate of 5.1 % (FAO, 2010). DFD is a nationwide and acute problem because forests and lands are the most important resources supporting livelihood through agriculture and wood-based incomes (Dourma et al., 2009; Wala et al., 2012). The most common threats to forest and land health are bushfire, excessive wood extraction, grazing and agriculture.

Regarding soil degradation as an indicator of LD, very few studies have been undertaken to assess soil degradation. At farm level, studies indicated that cultivation induced not only loss of vegetation cover but also soil nutrient depletion (Kintché et al., 2010; Sebastia et al., 2008). Soil erosion has also been mentioned as a major problem affecting steeper slopes in the northern regions (Poch & Ubalde, 2006). In addition, an indicator map of human-induced LD revealed many hotspots(Brabant et al., 1996). Like many of the landscapes in Togo, the Mo basin experiences high human-induced LD, likely to be worsened in forthcoming decades. However, to date, attempts of LD mitigation have not provided analysis of land dynamics in relation to soil conditions and the possible soil-vegetation interactions evolving the landscape patterns. In this regard, studies are needed for the Mo basin, which is deemed to have good potentials for landscape conservation(Dourma, 2008; Wala et al., 2012; Woegan, 2007).

1.2.2. Research approach to address multifaceted knowledge needs for sustainable land management and restoration in the Mo basin

In attempt to address LD in the Mo basin, the research approach covers the knowledge gaps and needs on the contemporary soil-landscape and vegetation patterns, the necessity of legacy information usage for landscape monitoring and the need of decision support tools for landscape planning and restoration.

First, understanding the level of and vulnerability to degradation that could be considered in planning and management options (Bennett et al., 2015; Hunke et al., 2015) requires sound assessments of the soil-vegetation interactions. Locally, most of the studies that have been conducted, have been directed at impact assessment (Fontodji et al., 2009; Kintché et al., 2010; Sebastia et al., 2008). Furthermore, understanding the ecosystem patterns and the level of human transformation help to develop novel strategies of monitoring (Turner, 2010; Verburg et al., 2013). The mountainous Mo basin, offer important capital for biodiversity conservation (Wala et al., 2012; Dourma et al., 2009) and potential for carbon sequestration (Folega et al., 2015), requiring an understanding the landscape heterogeneity and dynamics. Thus, the real potential of the contemporary patterns of soil-vegetation landscapes in response to multiple ecological, human and topographical factors still need investigation in terms of ecological values as well as the direct implication for local strategies of climate change mitigation.

Next, efficient strategies to turn over the alarming DFD and desertification require efforts to trace historical landscape patterns. As good proxy, multi-temporal satellite data provides an insight to assess the historical changes of landscape cover and provide basis for landscape planning in long-term perspectives (Tortora et al., 2015). In the multifunctional landscapes of the Mo basin, embedding a diversity of landscapes under different levels of degradation (Folega et al., 2015), this approach is particularly relevant for the identification of degradation hotspots, the assessment of potential impacts on ESS, and the land monitoring, conservation and restoration.

Finally, the propensity of multidisciplinary researches is their capability to investigate alternative options that could help in abating LD. In the multifunctional landscapes, typical coupled human-environment systems, integrated management tools have provided successful insights to support adapted landscape management (Le et al., 2010; Tamene et al., 2014). In this regard, this research is premised on the necessity of investigations towards the development of landscape planning and management tools in order to proactively analyse and strengthen the potential synergies for landscape conservation and restoration strategies in rural areas of Togo.

1.3. Research objectives

The aim of the research was to enhance knowledge on the spatio-temporal landscape patterns across the Mo River basin and propose supporting measures for LD mitigation and landscape restoration. Five specific objectives guided the research:

(i) To determine the patterns of soil conditions in relation to biophysical and human factors in the Mo River Basin;

(ii) To determine the vegetation patterns in relation to biophysical and human factors;

(iii) To assess the LUCC in the Mo basin over the period 1972-2014;

(iv) To assess the potential impacts of LUCC trajectories on the landscape configuration, soil erosion, soil organic carbon and total nitrogen;

(v) To model the landscape vulnerability to soil erosion and land management options in the Mo Basin.

1.4. Outline of the Thesis

Chapter 1 presents the general introduction, the problem statement, the objectives of the study and the corresponding research questions. Chapter 2 gives a background review of literature on the phenomenon of landscape dynamics and degradation as impacted by human pressures and natural processes. Chapter 3 to 7 report the methodology and discuss the research outputs in respect to each specific objective. The physical, socio-economical and human settings of the Mo River basin are presented along with these chapters. Chapter 8 presents the research findings in line with the specific objectives and makes appropriate recommendations.

CHAPTER 2: LITERATURE REVIEW

2.1. Concepts and definitions

Land, according to the United Nations Convention to Combat Desertification (UNCCD), is the "*terrestrial bio-productive system that comprises soil, vegetation, other biota, and the ecological and hydrological processes that operate within the system*". In this sense, land may refer to all earth components that sustain the provision of ESS at any a scale.

Landscape: According to Verburg et al. (2013), landscapes are the result of spatial heterogeneity in the physical environment and the interactions of humans with the environment. According to Antrop (2005), landscape is an integrative concept having the following characteristics: (i) a spatial entity (extent and scale) and territorial properties, (ii) is perceived and experienced, (iii) is heterogeneous and structured with a spatial organization and management that is largely influenced by humans, (iv) dynamic and changes are an inherent property of landscape.

Land cover, land use: according to the land cover classification system of the FAO "*land cover"* is the observed (bio) physical cover on the earth's surface while "*land use"* refers to the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it. In this sense, land cover may include vegetation, surface water, bare soil at any topographical location and human-made structures (paved areas, settlements, etc.).

Land use/cover change (LUCC) may refer to a temporary or permanent shift from one land use/cover type to another. Land cover conversion, the complete replacement of one cover type by another, can be separated from land-cover modification that refers to subtle changes that affect the character of the land-cover

(Braimoh, 2004). In this sense, temporary change may imply land cover modification whereas permanent change refers to land cover conversion.

Land degradation (LD), according to UNCCD, is the "*reduction or loss of the biological productivity, resulting from land uses, or combination of processes, such as soil erosion, deterioration of properties of soil and long-term loss of natural vegetation*". It involves all processes inducing the reduction or loss of ESS, i.e. a decline of land performance (MEA, 2005).

Desertification is common term used for LD in dryland areas and/or the irreversible change of the land to such a state it can no longer be recovered for its original use.

Soil erosion or *soil loss*, according to FAO, refers only to absolute soil losses in terms of topsoil and nutrients, and occurs as a natural process in mountainous areas, and worsened by poor land management practices.

Ecosystem services (ESS) are the direct and indirect contributions of ecosystems to human well-being. According to MEA (2005), ESS are the benefits people obtain from ecosystems, including soil stabilization, climate change mitigation, water conservation, and erosion and desertification control (FAO, 2010).

Spatially explicit modelling is the representation of a real world phenomenon that includes the geographic location of the system being modelled.

(Forest) Landscape restoration (FLR) is defined as a planned process to regain ecological integrity and enhance human well-being in degraded landscapes (ITTO & IUCN, 2005). Rather than aiming to restore forests to their original state, FLR is an approach that aims to strengthen the resilience of forest landscapes.

2.2. Phenomenon of land degradation and causes

2.2.1. Land degradation dimensions

With regard to the definition of LD mentioned in the previous section, the processes inducing LD include three major dimensions. First, LD is apparent through physical processes such soil erosion by water or wind and desertification, as a consequence of human impacts (Brunner et al., 2008; Le et al., 2012b; Tamene et al., 2006). Next, chemical quality changes such as soil acidification, leaching, and productivity loss due to nutrient depletion, are used as indicators of LD at agro-system levels (Waswa et al., 2013). LD can also be apparent when biological processes such as the reduction in land productivity and land biodiversity loss occur persistently in a given region (Traore et al., 2015; Vu et al., 2014). These three LD processes can occur separately or simultaneously, with multifold causes. LD is an ambiguous terminology and a very difficult concept to define since the boundaries between degradation and improvement are not clearly delineated (Laestadius et al., 2011). Based on the management and assessment perspectives, one can consider a certain situation of land cover as degraded, whereas another view can qualify the same state as an improvement situation.

2.2.2. Causes and assessment of land degradation

Approaches of LD assessment encompass short- and long-term analyses that should be integrated to achieve the monitoring and mitigation objectives. Early studies (Evans & Geerken, 2004; Wessels et al., 2007) corroborated by recent researches (Le et al., 2012b; Traore, 2015; Traore et al., 2015) highlighted the importance of the usage of multi-temporal inter-annual NDVI data and rainfall records for assessing LD. It is obvious that both climatic dynamics and human interference affect biomass productivity at large scale as well as local level. In separating the natural causes from the human-induced ones, LD could be detected to guide mitigation pathways at different scales. Furthermore, large scale projects such as Land degradation Assessment in Drylands (LADA), Global Assessment of Land and Soil Degradation (GLASOD), Global Land Project (GLP) assessed LD at large scales using various approaches. For instance, the World Overview of Conservation Approaches and Technologies (WOCAT) used expert judgments, questionnaire and historical earth observation data (NDVI time series).

However, such approaches have shown major limitations in terms of suitability for small-scale planning, the excessive usage of expert judgements and perceptions, and the abstraction of finer resolution of land conditions (Rhodes, 2014; Verburg et al., 2013). Addressing such limitations requires landscape level assessment of the phenomenon to arrive at real solutions based on local conditions and continuous assessment (Verburg et al., 2013). The integration of satellite images, geographic information systems (GIS), landscape metrics and field data have provided great insights in conducting LD assessment at small scales. While satellite images help in comprehending spatial and temporal dynamics of lands (Farooq, 2012; Rogan & Chen, 2004), the metrics are useful in assessing landscape patterns, biodiversity, and ecological sustainability (Renetzeder et al., 2010; Schindler et al., 2008; Schindler et al., 2013). Satellite observations have provided a successful and cost-effective approach in quantifying landscape spatial patterns and landscape ecological integrity, as well as assessing LD over different landscapes (Laestadius et al., 2011).

2.3. Land use/cover change and land degradation

2.3.1. Assessing LUCC for LD monitoring

Earth observation and GIS have helped in the development of various environmental management methodologies that are more advantageous with regard to the wide range of spatial and temporal coverages. Therefore, LUC mapping from the combination of classification techniques (supervised and unsupervised) offers advantage for monitoring landscape dynamics (Kuemmerle et al., 2006). For inaccessible areas, landscape information could be retrieved from such data and techniques (Cambule et al., 2013; Shrestha et al., 2014). Ground-based methods uniquely could not provide such advantages though they are important in supporting the space-borne observation of the Earth surface for effective LD assessment (Waswa et al., 2012). Satellite data are available from different sensors such as Landsat, Quickbird, MODIS, RapidEye, SPOT, etc. and used in different perspectives to monitor LD.

2.3.2. Land degradation and soil quality

Soil is a component of soil-landscape system, requiring much more attention to support landscape integrity, as it is a vital resource for food and fibre to support an increasing world population. Thus, soil quality has gained interest, as it is at the forefront of issues relating to environmental monitoring and food security (Oladele & Braimoh, 2011; Rhodes, 2014; Stockmann et al., 2015). Referring to the definition of soil quality related to agricultural productivity, indicators of soil quality refer to the soil properties influencing its potential to perform functions and provide services. The rapid degradation of soils, water and biodiversity in agricultural landscapes seriously compromises ESS in agricultural landscapes and reduces the resilience of food systems (MEA, 2005). However, far beyond the sole agricultural purpose, Parr et al. (1992) referred to soil quality as "*its (soil) capability to produce safe and nutritious foods and crops in a sustainable manner over the long term, and to enhance human wellbeing without adversely impairing the natural resource base or adversely affecting the environment*". The need for the environmental preservation aspect makes this definition interesting as it gives insight to the biogeochemical cycles involving climate change. In this study, emphasis is given to SOC and TN, and their spatial distribution in response to LUC types, landforms, biophysical settings and human interferences.

2.3.3. Distribution of SOC and TN, and its drivers

Naturally, the distribution of soil resources in terms of quality is erratic but highly dependent on the climatic, biophysical and other determinants of its formation and dynamics (Braimoh & Vlek, 2008; Brevik, 2013). As indicator of LD, SOC and TN are often used to weigh land health in comparison with benchmark values (Waswa et al., 2013). SOC and TN have been shown to be very sensitive to LUCC, as it causes their depletion through agricultural conversion (Abera & Belachew, 2011; Selassie et al., 2015; Were et al., 2015; Winowiecki et al., 2015). However, the nature of the impacts of shifting cultivation on soil properties in the tropics often depends on the cultivation system and the soil properties investigated (Ribeiro Filho et al., 2015). The greatest nutrient fluxes (especially SOC) between the atmosphere and the Earth's surface are mostly human-induced, especially LUCC (Traoré et al., 2015; Villarino et al., 2014). Agricultural deforestation is indicated to decrease SOC up to 63 % (Vågen et al., 2005) while LD in semi-arid lands could reduce the potential of soil CO_2 respiration up to 82 % (Traoré et al., 2015).

Landform has been shown to play a significant role in the spatial distribution of SOC and TN. In mountainous areas, LD as soil erosion and soil quality loss often emerges from the landform-induced processes such as surficial runoff, enhanced by land mismanagement options (Tamene et al., 2006; Tesfahunegn et al., 2014). Inland valleys and lowlands are commonly revealed to be nutrient-richer (Xue et al., 2013) as they receive much more sediments and matter flows from uplands. Therefore, it appears that topography, in combination with land use and other *in situ* conditions, is essential for a clear understanding of the interactions between landscape patterns and LD, especially in mountainous regions (Shrestha, 2015). As suggested by Rhodes (2014) and several other authors (Benin et al., 2011; Brevik, 2013; Lal, 2014; Vagen & Winowiecki, 2013), it is worthy to build knowledge on SOC and TN dynamics as influenced by land uses and natural factors for mitigating climate change and LD.

2.4. Vegetation patterns as impacted by various drivers

Various drivers inducing LUCC affect the vegetation pattern, which is one of the most important terrestrial carbon pools. However, human-appropriation of land resources indicated patterns that have exceeded the earth's production capacity over several decades (Le et al., 2012b). Yet, vegetation cover is a key component in climate regulation, and useful indicator for evaluating the geosphere-biosphere-atmosphere interaction (Salim et al., 2008). Agricultural share, wood industries, unsustainable wood-based product collection, and energy consumption are the most direct drivers of vegetation changes in landscapes (Chillo et al., 2015; Pravalie et al., 2014; Smith et al., 2014; Valle Junior et al., 2015). These increasing pressures have induced vegetation decline, the loss of native biodiversity, and affected carbon balance.

Besides, it has been shown that strong correlations exist between landscape patterns and complex biophysical conditions. Indeed, though Avohou and Sinsin (2009) reported that topography had no significant effects on plant biomass in the Atacora Mountains of Benin, it is indicated in several studies that landform and soil nutrients are among the most important factors of the distribution of plant ecological groups within a landscape (Adel et al., 2014; Dalle et al., 2014; Tavili & Jafari, 2009). Furthermore, any vegetation type is tied to particular soil conditions which highly vary spatially, even within the same cover type (Wiesmeier et al., 2014b). Accordingly, landscape positions and elevation induce the spatial variability of vegetation features. Based on these multiple findings, it is evident that topography in relation with soil conditions plays a determinant role in vegetation patterns (Aynekulu, 2011; Wala et al., 2012). In this respect, Tavili and Jafari (2009) indicated that an examination of these relationships is necessary for better land management. Meanwhile, using a combination of historical data and biophysical conditions, studies have revealed the negative influence of human disturbances on plant cover and forest ecosystems (Damnyag et al., 2013; Dewi et al., 2013; Folega et al., 2014b). With regard to these diverse findings on the biophysical determinants of landscape patterns and dynamics, landscape-specific studies are needed to guide decision making for adapted land use.

2.5. Land degradation and soil erosion

2.5.1. Soil erosion processes and factors

Depending on the erosive agent, there are two types of soil loss: erosion by water and by wind. However, this study focuses the water-induced soil loss. The latter is known under three forms: sheet, rill, and gully (Shoshany et al., 2013). Sheet and rill erosion refer to processes of soil particles detachment and transport by raindrops and overland flow. Soil erosion is therefore a form of LD (Martin-Fernandez & Martinez-Nunez, 2011). Soil loss occurs when erosion rates are greater than deposition rates, and its severity depends on many factors including natural processes and land management (Tamene & Vlek, 2007; Vlek et al., 2008). Tolerable soil loss varies according to regions and land use/ management purposes. It ranges from 2 – 3 Mg $ha^{-1}y^{-1}$ for agricultural fields to 12 -13 Mg $ha^{-1}y^{-1}$ or even 18 Mg $ha^{-1}y^{-1}$ for wild landscapes and mountainous tropical areas as in Kenya (Tamene, 2005; Tesfahunegn et al., 2014) or 15 Mg $ha^{-1}y^{-1}$ in West Africa (Le et al., 2012b; Roose, 1977). In certain SSA environments, tolerable soil loss rate has been exceeded, especially in highlands due to poor management (Le et al., 2012b; Tamene & Le, 2015).

The important factors influencing soil erosion patterns within a catchment include natural (e.g. landform, soil characteristics and rainfall) as well as human factors (e.g. land use, conservation practices) (Goldman et al., 1986; Lal, 1993). The soil type and landforms play major roles in the erosion processes, as they define soil erodibility and vulnerability to surface runoff. Studies also revealed that landforms and land use in highlands have been targeted as potential drivers of soil erosion in mountainous areas, suggesting that LUC types could have limited impacts on soil erosion provided that good management practices are developed (Labriere et al., 2015; Tesfahunegn et al., 2014; Zhou et al., 2014a). Accordingly, important negative correlations have been revealed between vegetation decline and soil erosion by water (Shrestha et al., 2014; Tamene et al., 2006). Vegetation cover is therefore crucial for the mitigation of soil erosion patterns and intensity (Morgan & Duzant, 2008). Some studies even showed that soil erosion is more sensitive to land use than rainfall intensity and landforms (Ouyang et al., 2010; Pacheco et al., 2014; Qiao et al., 2015).

2.5.2. Soil erosion measurement and modelling

The estimate of the magnitude and the spatial distribution of soil loss areas are commonly performed using various approaches spanning from plot measurements to spatially explicit models in relation to various social and biophysical factors (Tamene et al., 2015; Le et al., 2012b; Tamene et al., 2007). Though plot measurements provide good accuracy and precision to some extent, they are not only time consuming and costly, but also their spatial coverage is restricted to the experimental plots. Spatial-explicit models in relation with field observations, however, provide insights to large spatial coverage based on the principle of similar landscape conditions and similar environmental and hydrological processes (Tamene & Le, 2014).

Considering soil erosion as a phenomenon, it is represented through methods employing equations describing the link between environmental parameters that offer better explanations of the phenomenon (Rhodes, 2014). Depending on data availability challenges, soil erosion modelling is commonly assessed using different models. The most implemented is the Universal Soil Loss Equation (USLE) which was later modified several times as the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997). More RUSLE applications are evolving in large and complex landscapes (Zhou et al., 2014; Le et al., 2012b; Tamene et al., 2014; Tamene & Le, 2015). Furthermore, models such as Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998), Morgan-Morgan-Finney (MMF) (Morgan & Duzant, 2008; Morgan et al., 1984), Water Erosion Prediction Project (WEPP) (Flanagan & Nearing, 1995) have been proposed and used to represent the spatial distribution of the erosion phenomenon. However, most of these models have more specificity related to data-demand and environmental settings of the implementation area. Other distributed models such as Unit Stream Power-based Erosion/Deposition (USPED) (Mitasova et al., 1996) are GIS-based models with minimal input data.

Among these models, RUSLE and derivatives are widely used to model soil erosion severity patterns (indices of relative soil loss using severity classes rather than absolute values) (Ashiagbor et al., 2013; Fathizad et al., 2014; Guo et al., 2013; Le et al., 2012b; Tamene et al., 2014; Zhou et al., 2014c). Its advantage resides in its low input parameters and it can be run in various modelling environments. An example of successful application is the Landscape Planning and Management Tool (LAPMAT), a spatially distributed RUSLE-based model that offers a capability to integrate and evaluate soil loss and assess sediment delivery ratio (Tamene & Le, 2014). However, concerns exist about the use (or misuse) of the RUSLE regarding the validity of its predictions (Rhodes, 2014). As model validity can be based on the construct process i.e.

related to the validity of its input parameters, it is therefore relevant to ensure the reliability of the inputs during the RUSLE-based soil erosion modelling.

2.6. Approaches for modelling ESS dynamics

2.6.1. Complexity in modelling ESS

The complexity in understanding the dynamics of ESS remains a major challenge due to rapid and unexpected land transformation as well as changes in local landscape conditions (Bai et al., 2013), calling for methods and models for large-scale transdisciplinary analyses. As indicated by Turner et al. (2015), there is a need for further development of integrated approaches, which consider all four types of capitals (human, built, natural, and social), and their interaction at spatially explicit, multiple scales. The spatial explicit approaches targeting small to medium scale showed evidence in supporting outputs to break down such complexity (Tamene et al., 2014; Le et al., 2012b). The complexity also resides in the coupling of all capitals into approaches and systems (Liu et al., 2007), and in the consideration of feedback effects of system dynamics (Huber et al., 2013; Le et al., 2012a). Integrated tools for management of land resources such as Multi-Agent Systems, Human-Environment System, and Agent-Based Modelling appeared to be worthy paths for addressing LD.

2.6.2. Necessity of spatially explicit modelling for LD mitigation

The consideration of space in modelling of landscape processes offers the advantage of integrating heterogeneity according to location (Jepsen, 2004; Le, 2005) but also spatial dependence and self-organisation of ecological processes comprising human dimension (Verburg et al., 2013). For instance, soil erosion is driven by natural process constrained by biophysical factors as well as LUCC that vary significantly across space. Spatially explicit models (SEMs) allow better understanding of the spatial patterns of factors of the phenomenon being modelled (Laue & Arima, 2015; Valbuena et al., 2010). Case applications of SEM are the identification of hotspot areas of LD to support adaptation and mitigation strategies (Le et al., 2012b; Tamene et al., 2014). A direct implication of SEM is their ability to derive information on remote areas and scale out results to other parts of landscape, especially for inaccessible lands. Accordingly, Tamene and Le (2015) used the concept of similar environmental constraint envelop (SECE) for SEM to propose soil loss patterns in SSA.

Despite the several interests of SEMs, caveats reside in the consideration of such spatial heterogeneity through earth observation and the representation of the different interactions of ecological processes. It cautions about the high heterogeneity residing among the provided ESS by the LUC units as obtained from remote sensing, which is also dependent on the spatial resolution (Verburg et al., 2013). Indeed, it is possible, if not real that same LUC types offer different services depending on *in situ* conditions. The abstraction of such variability in SEMs may be considered as limitations to such models. In this regard, the concept of SECE (Tamene and Le, 2015) have certain limitations as the methods of determining SECEs might offer biases based on *in situ* site-specific conditions offering different perspectives. Next, the interactions between components of ecological processes and human interferences still evolve new concerns since such interactions are difficult to be considered in SEMs. Yet, SEMs have gained more attention in investigating the patterns of natural phenomena such as soil erosion (Mondini et al., 2012; Wasige, 2013; Yadav & Malanson, 2013).

2.7. Landscape restoration and implications for climate change

More than 80 % of the land surface is directly affected by human activities while the remainder is indirectly affected through human impacts on climate, air quality, water quality and flow changes (Ellis, 2011; Foley et al., 2005). The biomass productivity based mapping of LD showed that about 29% of global land area in all agro-ecologies and LUC types (Le et al., 2012b) were degraded while over 65 % of agricultural lands experienced LD (Vlek et al., 2008). In the savannah zone of West Africa, due to the increasing needs for land restoration to support agrosystems and climate mitigation, bush fallow have been shown to marginally compensate LD caused by farming systems (Jamala & Oke, 2013). Efforts to assist recovery of land ecosystems are necessary and require an understanding of the complex factors and actors behind the degradation processes. Thus, Mutoko et al. (2014) suggest that the spatial identification of the resource-dependency is important to capture the extent and context of interventions.

In agricultural landscapes, the existence of non-attractive policy frameworks, the lack of regulatory policy at local level, the weak land tenure, and other socio-economic factors have been mentioned as factors hindering the adoption of land conservation measures (Cordingley et al., 2015; Rosenstock et al., 2014; Shepherd et al., 2014). In this context, it is suggested that incentives can encourage successful and sustainable management of soil fertility (Kassie et al., 2015; Marenya et al., 2012) Furthermore, Nkonya et al. (2011) found a strong linkage between poverty indicators and LD in SSA, suggesting that landscape restoration should target poverty root to make efficient strategies of restoration. In the scope of adopting and perpetuating SLM for farmer resilience to climate change and food security, Mutoko et al., (2014) found that SLM is seen to be fostered through the implication of all stakeholders to ensure effective implementation approaches and methods at different levels (Coe et al., 2014; Nkonya et al., 2011). Landscape restoration through sustainable agroforestry is also seen to offer great potential for achieving sustainable land development and climate change mitigation (Mbow et al, 2014a; Mbow et al, 2014b).

CHAPTER 3: STORAGE AND DRIVERS OF SOIL ORGANIC CARBON AND TOTAL NITROGEN IN MO LANDSCAPES [1]

3.1. Introduction

In Sub-Saharan Africa (SSA), land management approaches induce land degradation (LD) and soil quality loss through the poor resource allocation and use, inappropriate land-related policy development and inadequate planning and management strategies (Petter et al., 2012; Portman, 2013; Primdahl et al., 2013). As indicators of soil performance and productivity, soil nutrients, especially SOC and TN, provide information on land health (Kintché et al., 2010; Touré et al., 2013; Vagen & Winowiecki, 2013; Wiesmeier et al., 2014a; Xiong et al., 2014; Zucca et al., 2013). Though SOC and TN are not the sole important elements for soil fertility and productivity measurement, they are increasingly recognised as major contributors to biogeochemical cycles and climate change mitigation processes. However, land use/management (e.g. cropping, grazing and mining), and environmental factors (climatic, edaphic, etc.) have been targeted as major factors affecting soil system in the biogeochemical cycles (Dorji et al., 2014; Gutiérrez-Girón et al., 2015; McGranahan et al., 2013; Vagen & Winowiecki, 2013; Villarino et al., 2014). Foremost of the concerns in land management is the inefficiency of the current traditional farming systems attributed mainly to the degradation of soil quality through organic matter (OM) depletion, productivity decline, and soil erosion (Sebastia et al., 2008; Touré et al., 2013; Vagen & Winowiecki, 2013). They require great attention as they play a key role in controlling soil chemical amounts and distribution (Biro et al., 2013; Houghton & Goodale, 2004) as well as carbon and nitrogen fluxes (Selassie et al., 2015; Tanner et al., 2014; Touré et al., 2013; Xue et al., 2013).

[1] This chapter is a revised version of a manuscript under review in *Ecological Engineering*.

In the Mo basin, little is known about the controlling biophysical factors of the SOC and TN contents at the landscape level. Research in this area is needed to support sustainable land management in multifunctional landscapes. Knowledge on SOC and TN cycling in different ecosystems is equally crucial in understanding their contribution to the viability of climate mitigation and landscape restoration strategies. In contributing to filling this gap, this study was carried out to determine the spatial patterns of the soil conditions in relation with biophysical and human factors in the multifunctional landscapes of Mo river basin (Togo). The focus was the quantification and distribution of the potential storages of SOC and TN up to 30 cm depth in relation to vegetation types, landscape positions and land protection regime. The relationships between soil chemical conditions, and *in situ* ecological and biophysical conditions were investigated and their implications for sustainable landscape management and climate change mitigation were discussed.

3.2. Material and methods

3.2.1. Study area

a) Geographical location of the Mo river basin

The study was carried out at specific sites in the Mo river basin. Figure 3.1 shows that Mo River basin is located in the central part of Togo. It is a sub-unit of the Volta River Basin which covers about 400,000 km^2 within the West African savannah zone. Located between Latitudes 8^{O} 45′ and 9^{O} 30 ′ N, and Longitudes 0^{O} 30 ′ and 1^{O} 30 ′ E, the Mo River basin covers a total area of about 1,485.92 km^2. About 46 % of the Mo landscapes fall under three different protected areas. Four riparian prefectures (Tchaoudjo, Bassar, Sotouboua and Plaine de Mo) share the Mo River basin.

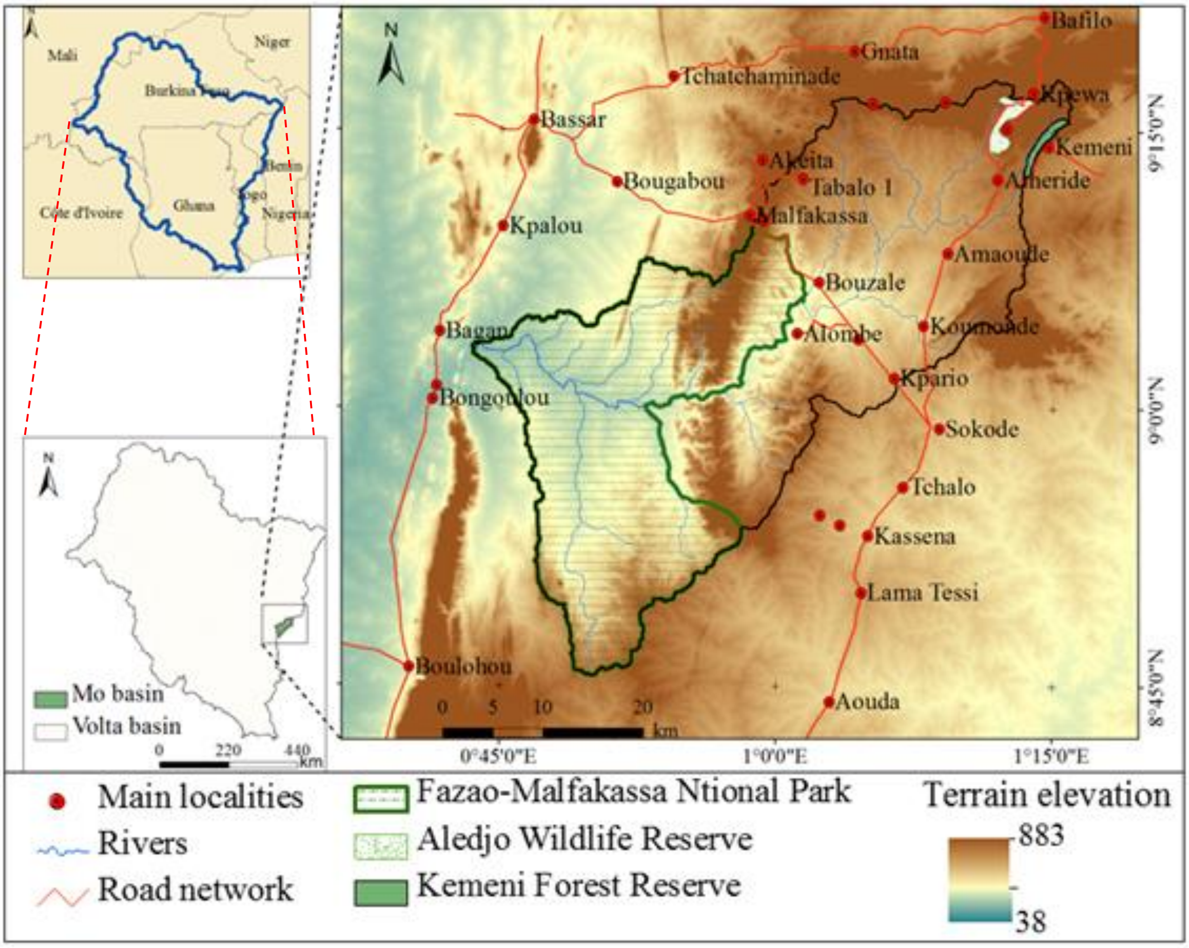


Figure 3.1. Location of the study area

b) Climate and climate change

The climate is tropical sub-humid characterised by a rainy season from April to October (Petit, 1981). Mean annual rainfall is between 1200 - 1300 mm with an irregular spatial-temporal distribution. Average minimal and maximal temperatures reach 19ºC in January with the Harmattan and 30ºC in April, respectively. Evapotranspiration is generally high, especially during the dry season and can reach 1600 mm per annum. The climate at the national level is projected to experience a rise northward of about 0.8 to 1.0 ºC and 2.3 to 2.7 ºC in average temperatures by 2025 and 2100, respectively. At the same time, a decline in precipitation is estimated to about 0.3 % by 2025 and 1.25 % by 2100 (MERF-Togo, 2010)

c) Relief and hydrography

The dominant geomorphological unit in the Mo basin is the Atakora mount chains. The altitude reaches 800 m, especially in Aledjo Mountains (Figure 3.2). The mountains are of variable heights, including Mazela (704 m), Akitili (861 m), Kouzé (625 m) and Kpeya (652 m). The Malfakassa Mountains comprise Ouassi (568 m), Zandebou, Tchakouya, Timbou and Balankan.

The drainage density is high with Mo, Loukoulou, Kamasse, and Bouzalo as the most important streams of the basin. Mo River and its tributaries dominate the stream network. The river flows in the east-west direction across the Ghana-Togo boarder in the Volta River. It is a sub-basin of the Oti River basin, a unit of the Volta watershed. A large part of the stream network experience low water level during the dry season. Owing to its location in mountainous areas, the Mo River basin experiences flash floods as well as some seasonal cascades.

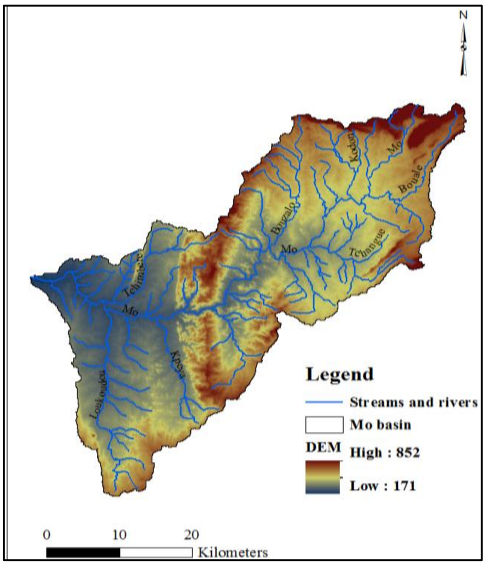


Figure 3.2. Terrain elevation and river network in the Mo basin

3.2.2. Sampling design and soil sample collection

Fieldwork was undertaken for soil sample collection in the different LUC types occurring at different landscape positions. Since topography was the main constraint during the fieldwork, plots were installed randomly along a topographical gradient from valleys to top-slopes (Wala et al., 2012). In addition to physical conditions (vegetation cover and accessibility), sampling was done following the land protection regime (PA versus UPA) to assess the effect of conservation on SOC and TN. Thus, disturbed soil samples were taken at different sites according to accessibility and vegetation homogeneity in such a way to represent the various vegetation types under both PA and UPA in the landscape. The collection of each sample was set in areas where vegetation features were homogeneous over a minimum area of 1 hectare (100 m x 100 m). The geographic coordinates of each core site was recorded using a GPS handheld sensor. In total, seventy-five sampling sites were investigated in the different cover and management types including farmlands (both cultivated and fallow). The samples were collected at two depths: 0 - 10 cm and 10 - 30 cm, hereafter named as topsoil and subsoil, respectively. At each site and for each depth, a composite sample was collected from five (5) random sites within a minimum area of 20 m x 20 m.

3.2.3. Laboratory analyses of soil samples

For laboratory analysis, the disturbed soil samples were air- and oven-dried, sieved through a 2 mm sieve before analysing for pH_{water}, SOC and TN content (in %). These elements are important indicators of soil productivity for crop production or natural ecological processes. Analyses were carried out on the samples for the topsoil and subsoil. No undisturbed soil sample was collected for the bulk density analysis.

The pH of the soil is an indicator of soil acidification, which depends on the soil parent material, the soil leaching in the environment and the land use/management. It was determined through a suspension of soil in 0.01M $CaCl_2$ solution in a 1/2.5 soil to solution ratio. A pH-meter and a combined glass electrode determined the pH.

SOC content was determined using the method of Walkley and Black (1934). The method consists of oxidising the OM by a known concentration of potassium dichromate (1.0N) solution added in excess. Using a diphenylamine as indicator in redox reaction, the excess unreacted potassium dichromate is titrated with Ammonium iron (II) Sulphate solution (0.5N). SOC is calculated as indicated in equation 3.1.

$$\% \, SOC = \frac{M(V_1 - V_2) * 0.39}{S} \qquad \text{(Equation 3.1)}$$

where M = molarity of Potassium dichromate; V_1= blank titration; V_2 = sample titration; S = weight of soil sample; and 0.39 is the constant to express the incomplete combustion of OM in the process.

TN was analyzed using the Kjeldahl method (Hanotiaux et al., 1975). It consists of boiling a homogeneous sample in concentrated sulfuric acid in order to transform the organic nitrogen into ammonia (NH_4) in an ammonium sulphate solution. Excess base (NaOH 10N) is added to the digestion product to convert NH_4 into NH_3, which is recovered by distillation in boric acid solution. Direct titration method is then used to quantify the amount of ammonia in the receiving solution using sulfuric acid N/50. The total nitrogen is then calculated using equation 3.2.

$$\% \, TN = \frac{x \, moles}{1000 \, cm^3} \frac{(V_s - V_b) \, cm^3}{mg} \frac{14 \, g}{moles} * 100 \qquad \text{(Equation 3.2)}$$

where V_s and V_b are the titration volumes of the sample and blank, and 14 g is the molecular weight of nitrogen (N).

3.2.4. Collection of *in-situ* ecological and other environmental variables

Ecological features and human disturbances were recorded at each sampling site. First, vegetation canopy cover, indicating the surface covered by the vertical projection of all tree foliage present in a given plot, was recorded by the Braun-Blanquet method

(Dimobe et al., 2014; Folega et al., 2012; Okou et al., 2014; Wala et al., 2012). Then, based on the occurrence (presence/absence) without any intensity gradient, the footprints of tree logging, cattle grazing, and wildfire were recorded as human disturbances. Finally, soil waterlogging was recorded as presence/absence data. In farmlands and fallows, supplementary features such as crop type and fallow age were noted but were not considered in data interpretation. These records were not available for all sites since land users were not around to confirm the guesses.

Other potential environmental parameters were extracted from Shuttle Radar Terrain Mission (SRTM) Digital Elevation Model (DEM) at 1 arc-second resolution (approx. 30 m) available from https://earthexplorer.usgs.gov. These parameters included different terrain attributes such as slope, Topographic Position Index (TPI), Stream Power Index (SPI), Topographic Wetness Index (TWI), mean altitude above channel level (Alt.ch), upslope contributing area (CA). TPI indicates the topographical slope position whereas TWI is a topographic indicator of spatial distribution of soil moisture conditions (Sørensen et al., 2006). SPI, CA and Alt.ch are used to indicate the potential effect of channel network on water flow, as indicators of soil drainage and its influence on soil chemical contents. All these topographic indices were used as potential indicators of the influence of geomorphic positions and hydrological processes on SOC and TN contents at the landscape level.

The potential effects of soil erosion were analysed by integrating the factors of the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997). In this study, the RUSLE-based factors used were the soil erodibility index (K factor), the rainfall erosivity (R-factor), and the vegetation cover index (C-factor).

The R factor was derived from Equation 3.3 using the average annual precipitation data, covering 16 regular gridded weather stations, downloaded from the

Global Weather Data (http://globalweather.tamu.edu/) (Dile & Srinivasan, 2014; Fuka et al., 2014). The Equation 3.3 (Roose, 1977) was successfully used in West African environments to calculate R-factor (Le et al., 2012b; Tamene & Le, 2015).

$$R = 0.577\,\mathrm{Pa} - 5.766 \qquad \text{(Equation 3.3)}$$

where R = annual rainfall erosivity (MJ mm/ha/h/y), and Pa = average annual precipitation (mm) of nearby stations.

The values for the K factor (in Mg ha^{-1}) were derived from Le et al. (2012b) in accordance with the dominant soil types from the Harmonized World Soil Database (HWSD) (FAO/IIASA/ISRIC/ISS-CAS/JRC, 2008). The two dominant soil types of the Mo River basin are Lixisols and Leptosols. The derived values for K factor were of 0.09 for Lixisols and 0.19 for Leptosols (Le et al., 2012b). The lack of experimental data for Mo basin to calculate K factor (Angima et al., 2003; Renard et al., 1997) constrained the estimation process.

The surface cover (C factor) as a factor of soil erosion potential was estimated based on the usage of satellite image as good proxy of land use/cover (LUC). Therefore, the C factor values were computed using the normalised difference vegetation index (NDVI) data of the Landsat 8 image of 2014 (http://www.earthexplorer.usgs.gov) using equation 3.4 (Le et al., 2012b; Parveen & Kumar, 2012; Tamene et al., 2014).

$$C = \exp\left[-\alpha * \frac{\mathrm{NDVI}}{(1-\mathrm{NDVI})}\right] \qquad \text{(Equation 3.4)}$$

where NDVI is the normalised difference vegetation index; α is a unit-less parameter determining the shape of the curve-relating NDVI and the C factor. The commonly used α-value of 2.5 (Le et al., 2012b) since it showed good agreement when compared with C-factor values derived from LUC data of similar studies.

All the spatial explicit variables potential controlling parameters of SOC and TN storages were extracted at 30 m-resolution. The various maps were then exported to a GIS for extracting values of the variables to GPS coordinates of the sample sites.

3.2.5. Mapping of the spatial patterns of SOC and TN contents

Spatial patterns of SOC and TN contents were developed based on the LUC types. Accordingly, the contemporary LUC types (Chapter 4 and 5) were used to map the average values of the measured SOC and TN. Based on these LUC types, and assuming similar soil cover to contain similar OM quantity, average SOC and TN for savannahs were obtained from the arithmetic mean of the records in shrubs and woody savannahs. Similar arithmetic averages were calculated on data from dry and riparian forests to get SOC and TN values for forest cover type whereas the values for the cultivated lands were obtained from farms and fallows. No data was set for water and settlements because soil samples were not collected in those LUC types.

3.2.6. Statistical analyses

All the investigated sites were described and compared using descriptive statistics. One-way analyses of variance (ANOVA at $p = 0.05$) were performed to evaluate the the difference of SOC and TN according to the four main factors (LUC types, topography, soil depths and land protection status). The post-hoc comparison of Tuker tests was used to detect least significance differences to support the ANOVA. Correlations between soil chemical properties and environmental variables were tested using pairwise correlation adjusted to Tukey significance level at 95% Confidence Interval. Multivariate approaches were used to identify the different relationships between environmental conditions and soil chemical properties at landscape level. A Canonical Correspondence Analysis (CCA) was performed to detect the effects of environmental variables on soil chemical contents for all investigated sites. Another CCA was used to explore the relationships between the distribution of soil parameters and ecological variables prevailing in agrosystems. Among all ecological features and human disturbances indicators used as explanatory variables, fire occurrence, farming or fallowing of the land, tree logging, soil drainage, protection status of lands, and cattle grazing were coded as dichotomous variables (0 = absence and 1 = presence). Data on topography, canopy cover and altitude above sea level were considered as simple variables at each site level. The matching similarity measure along a dendrogram was used to test the similarity/dissimilarity level among sites in agro-systems and discriminate different groups among agro-system sites based on the biophysical and nutrient content conditions. Orthogonal rotated loadings of principal factor analyses with significance level of 95 % CI were used to reveal and extract the main factors explaining SOC and TN distribution. These supplementary statistics were carried out to overcome the interrelationships between hypothesized environmental and topographical factors that control SOC and TN storage. For handling purposes, SOC for the topsoil (0 – 10 cm), subsoil (10 – 30 cm) and whole depth (0 – 30 cm) were coded as SOC10, SOC20 and SOC30, referring to the layer thickness. The same codification was applied in the case of pH and TN.

3.3. Results

3.3.1. Variations of SOC and TN under different land use/cover types

Total nitrogen contents in topsoil layer (0 - 10 cm) differed significantly ($p < 0.01$) under different vegetation types, both natural and cultivated lands (Table 3.1). The mean TN varied from 0.056 % in fallows to 0.159 % in dry forests. The lowest TN (0.018 %) was found in farms whereas the highest (0.378 %) was in dry forests. Agricultural landscapes exhibited the lowest TN stocks of 0.056 and 0.062 % in fallows and farms, respectively. Similar to the TN content in the topsoil, subsoil TN (10 - 30 cm) differed significantly according to LUC types ($p < 0.01$). The lowest (0.043 %) and highest (0.091 %) TN mean values occurred in the subsoil of woody savannahs and dry forests, respectively. At site level, records of TN ranged from 0.017 % to 0.191 % in the same cover types, respectively. Accordingly, cumulative TN (0 – 30 cm) varied significantly ($p = 0.01$) among the soil cover types. At this depth, the lowest mean value (0.098 %) occurred in fallows whereas dry forests had the highest mean TN content (0.248 %). In general, the results showed TN to be highly influenced by LUC types, with soils under moderate to high canopy cover being TN-richer. TN decreased with increasing soil depth, indicating that deeper soil layer contains less TN. At both depths, farm soils contained more TN than fallows, suggesting that fallows which are old farms had their soils highly depleted in TN during cultivation and have not had the requisite fallow duration for nutrient recycling and replenishment.

As far as SOC is concerned, a significant difference was observed in the topsoil according to the vegetation types ($p < 0.05$) (Table 3.2). The average SOC ranged from 1.81 % in fallows to 3.58 % in dry forests, which stored by far the highest SOC. Dry forests, gallery forests, and woodlands were significantly SOC-richer whilst man-made ecosystems were poor ($p = 0.026$). The magnitude of SOC was in the order of dry forests > gallery forests > woodlands > woody savannahs > farms > shrubs > fallows. In contrast, SOC in subsoil layer did not show significant differences in relation with the cover types (Table 3.2). However, the lowest (1.52 %) and highest (2.23 %) average content of SOC occurred in farmlands and woodlands, respectively. At site level, records ranged from 0.69 % in woody savannahs to 3.87 % in shrubs. SOC ranked as woodlands > fallows > dry forests > gallery forests > shrubs > woody savannahs > farms. The cumulative amount of SOC (0 – 30 cm) varied significantly ($p = 0.007$)

between the different LUC types. The lowest mean value (1.81 %) occurred in woody savannahs whereas dry forests stored the highest mean SOC (7.96 %).

In general, SOC is highly affected by land use in the topsoil layer whilst SOC variability in subsoil layer is not significantly related to land use and management. Soils under moderate to high canopy cover were richer in SOC. For all LUC types, SOC decreased with increasing soil depth. In the topsoil, farmlands contained more SOC than fallows, whereas the subsoil OM was lower in the fallows, suggesting that topsoil SOC significantly decreased during cultivation.

Table 3.1. TN (in %) for different land use/cover types and soil depths

Vegetation types	Mean ± StdDev	Min.	Max.	Skewness	Kurtosis	Tukey
	Topsoil (0 – 10 cm); ANOVA * (p = 0.000 at 95 % CI)**					
Dry forests	0.157 ± 0.11	0.053	0.378	1.25	0.50	A
Fallows	0.056 ± 0.03	0.040	0.104	1.90	3.38	B
Cultivating farms	0.062 ± 0.05	0.018	0.202	1.97	4.60	C
Gallery forests	0.159 ± 0.08	0.049	0.301	0.38	-1.11	AC
Shrubs	0.077 ± 0.04	0.033	0.157	1.35	2.19	BC
Woodlands	0.112 ± 0.05	0.047	0.239	1.19	1.90	AB
Woody savannahs	0.082 ± 0.04	0.035	0.146	0.52	-0.80	BC
	Subsoil (10 – 30 cm); ANOVA * (p = 0.002 at 95 % CI)**					
Dry forests	0.091 ± 0.06	0.042	0.191	1.14	-0.27	A
Fallows	0.043 ± 0.02	0.027	0.075	1.32	1.19	B
Cultivating farms	0.049 ± 0.03	0.019	0.147	2.40	7.29	B
Gallery forests	0.084 ± 0.03	0.045	0.124	0.05	-1.47	A
Shrubs	0.051 ± 0.01	0.040	0.073	1.63	2.99	B
Woodlands	0.066 ± 0.02	0.021	0.099	-0.41	0.31	AB
Woody savannahs	0.051 ± 0.03	0.017	0.093	0.64	-0.58	AB
	Whole depth (0 – 30 cm); ANOVA * (p = 0.000 at 95 % CI)**					
Dry forests	0.248 ± 0.15	0.116	0.569	1.48	1.60	A
Fallows	0.098 ± 0.04	0.069	0.178	1.72	2.60	BC
Cultivating farms	0.111 ± 0.07	0.043	0.283	1.67	2.40	C
Gallery forests	0.244 ± 0.11	0.109	0.422	0.35	-1.07	A
Shrubs	0.128 ± 0.05	0.073	0.230	1.51	2.83	BC
Woodlands	0.179 ± 0.06	0.116	0.325	1.75	4.21	AB
Woody savannahs	0.133 ± 0.06	0.061	0.239	0.81	0.05	BC

Note: *StdDev = Standard deviation; Min = minimum; Max = maximum; ANOVA = analysis of variance; Tukey = post-hoc multiple comparison using Tukey method.* ***Skew****. = Coefficient of Skewness;* ***Kurt****. = Coefficient of Kurtosis.*
**** = statistical significance at $p < 0.05$ (95% CI);* ***CI*** *= Confidence interval;* ***p*** *= probability value of the ANOVA tests ($p = 0.05$). Means that do not share a letter are significantly different; Capitalized letters stand for the outputs of post hoc tests at 95% CI.*

Table 3.2 SOC (in %) for different land use cover types and soil depths

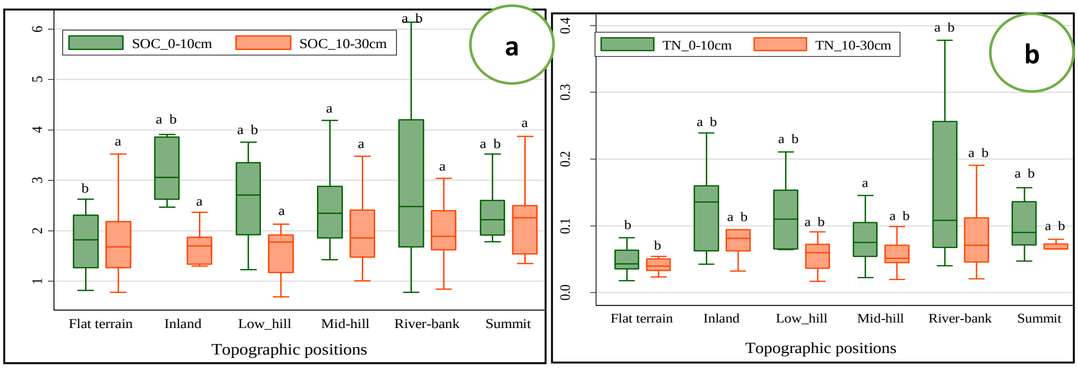
Vegetation types	Mean ± StdDev	Min	Max	Skewness	Kurtosis	Tuker
Topsoil (0 – 10 cm); ANOVA *(p=0.026 at α=0.05 CI)**						
Dry forests	3.58 ± 0.49	1.98	6.14	0.73	-0.52	A
Fallows	1.81 ± 0.79	0.82	3.03	0.69	2.25	AB
Cultivating farms	2.26 ± 1.00	1.27	5.50	2.33	6.20	AB
Gallery forests	2.86 ± 1.24	0.78	4.70	-0.32	-1.16	B
Shrubs	2.12 ± 0.62	1.05	2.63	-1.14	-0.14	B
Woodlands	2.71 ± 0.76	1.49	3.86	0.05	-0.77	B
Woody savannahs	2.33 ± 1.03	1.03	4.72	1.29	2.77	B
Subsoil (10 – 30 cm); ANOVA *(p=0.053 at α=0.05 CI)						
Dry forests	2.14 ± 0.49	1.33	2.80	-0.06	-0.84	
Fallows	2.20 ± 0.77	1.68	3.52	1.84	3.85	
Cultivating farms	1.52 ± 0.31	1.01	2.18	0.44	-0.18	
Gallery forests	1.93 ± 0.66	0.84	3.04	0.25	-0.47	
Shrubs	1.92 ± 0.92	1.18	3.87	1.95	4.16	
Woodlands	2.23 ± 0.63	1.01	3.48	0.19	1.09	
Woody savannahs	1.63 ± 0.89	0.69	3.19	0.87	-0.50	
Whole depth (0 – 30 cm); ANOVA *(p=0.007 at α=0.05 CI)**						
Dry forests	5.71 ± 1.44	4.34	7.96	0.54	-1.59	A
Fallows	4.01 ± 0.62	3.43	4.88	0.50	-1.19	AB
Cultivating farms	3.78 ± 1.06	2.61	7.09	1.89	4.99	AB
Gallery forests	4.79 ± 1.38	2.91	7.65	0.34	-0.12	BC
Shrubs	4.05 ± 1.20	2.69	6.47	1.52	3.10	BC
Woodlands	4.93 ± 0.93	3.47	6.36	-0.08	-1.18	BC
Woody savannahs	3.96 ± 1.75	1.81	7.71	0.90	1.24	C

***Note:** StdDev = Standard deviation; Min = minimum; Max = maximum; ANOVA = analysis of variance; Tukey = post-hoc multiple comparison using Tukey method. **Skew**. = Coefficient of Skewness; **Kurt**. = Coefficient of Kurtosis.*
**** = statistical significance at p < 0.05; **CI** = Confidence interval; **p** = probability value of the ANOVA tests (p = 0.05). Means that do not share a letter are significantly different; Capitalized letters stand for the outputs of Post hoc tests at 95% CI.*

3.3.2. Distribution of SOC and TN according to topographical positions

Figure 3.3a shows significant variation of SOC in relation with topography in Mo landscapes. Mean SOC values showed highest records in inland valley and low-slope. Flat terrain (1.89 %) and riverbank (2.16 %) exhibited the lowest SOC whereas low-slope (2.63 %) and mid-slope (3.53 %) had the highest SOC. The general trend was inland valley > low-slope > riverbank > mid-slope > top-slope > flat terrain. In the subsoil, no significant difference was observed. However, the riverbanks (2.06 %) and top-slope (2.03 %) recorded the highest average SOC. In this subsoil, the trend in SOC variability was top-slope > riverbank > mid-slope > low-slope > inland valley > flat terrain. In aggregate, SOC of the 0-30 cm depth was significantly affected by topography with mid-slope containing more SOC (5.42 %) than riverbank (4.22 %), low-slope (4.10 %), and flat terrain (3.84 %). The topsoil exhibited the highest SOC concentration for all the topographical positions while flat terrain showed the lowest mean records regardless of the depth.

As far as TN is concerned, on the average, its content in the topsoil varied significantly from 0.05 % on flat terrains to 0.150 % on mid-slopes (Figure 3.3b). The lowest record (0.018 %) was found in cultivated farms whereas the highest (0.378 %) was on mid-slope. TN in the topsoil ranked as inland valley > low-slope > riverbank > top-slope > mid-slope > flat terrain. In the subsoil, significant variations were observed according to topographical locations ($p = 0.002$). The lowest (0.046 %) and highest (0.084 %) mean values of TN were obtained for flat terrain and mid-slope, respectively. The records ranged from 0.017 % in flat terrain to 0.190 % in mid-slope. In contrast to the topsoil, the rank order was inland valleys > riverbank > low-slope > top-slope > mid-slope > flat terrain. For the total depth (0 – 30 cm), TN content varied significantly with lowest average content (0.097 %) in flat terrain and the highest mean record (0.234 %) on mid-slope. Meanwhile, the minimum and maximum of records were of 0.042 % and 0.569 % in the same topographical positions, respectively. The highest TN contents in both soil depths were recorded in inland valley, riverbank, and low-slope.



Note: *Shared letters are post-hoc Tukey test for group discrimination. Values with same letters belong to the same group (i.e. not significantly different at 95% CI).*

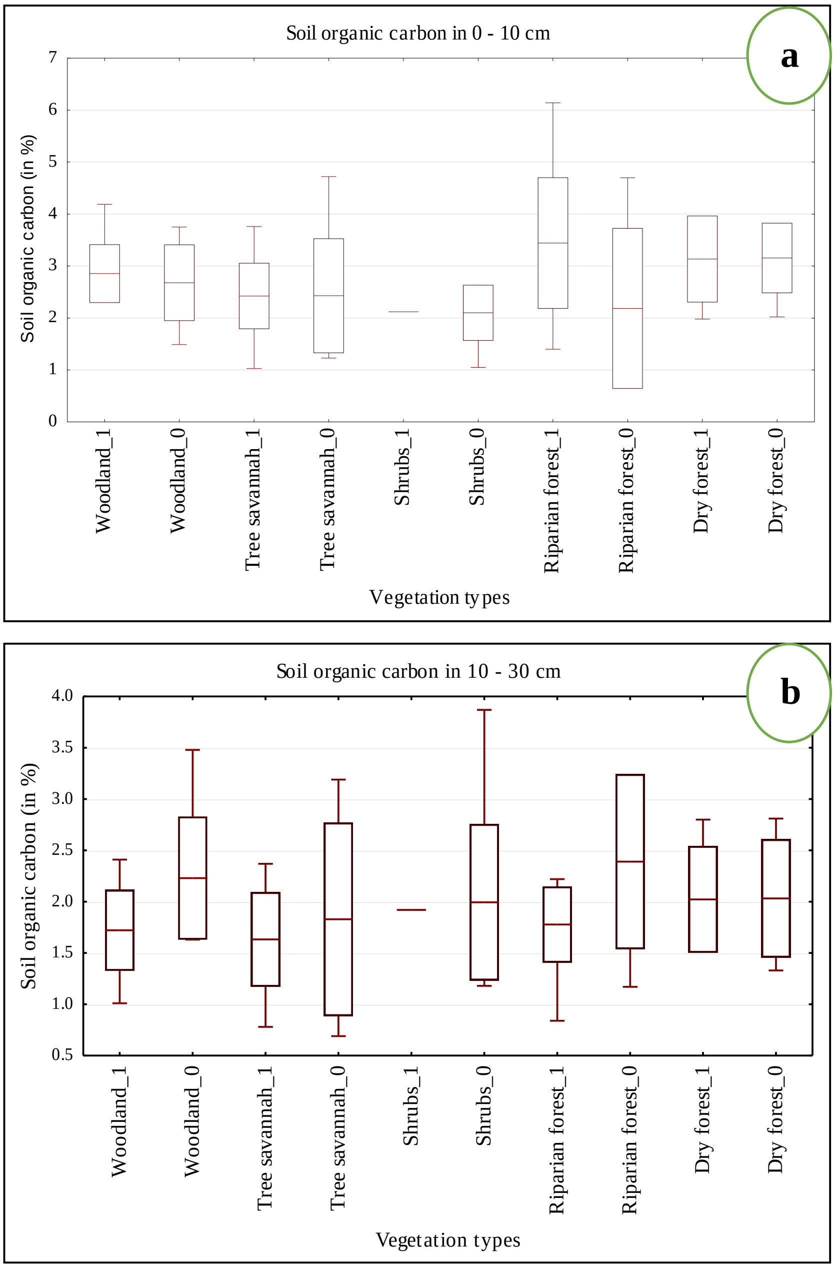
Figure 3.3. SOC (a) and TN (b) according to different landscape positions

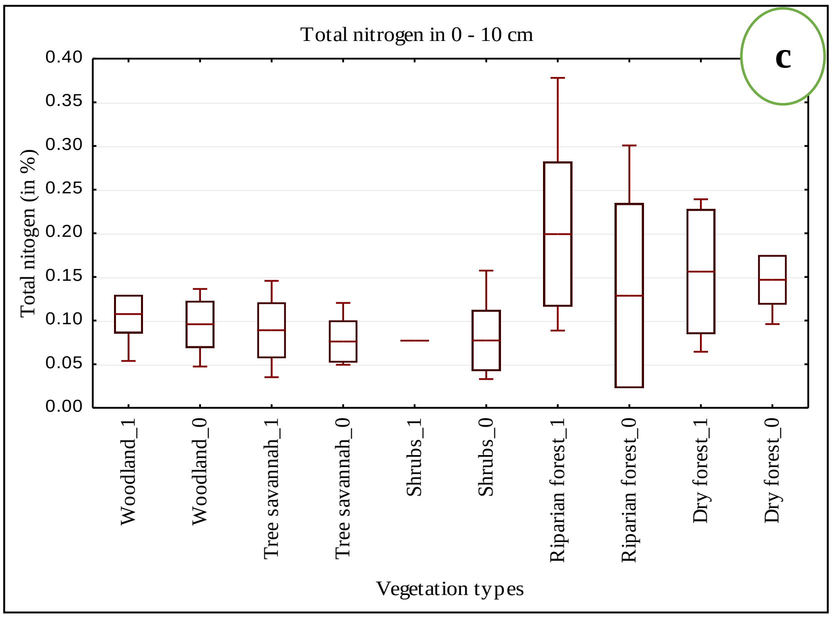
3.3.3. Influence of land protection regime on soil chemical properties

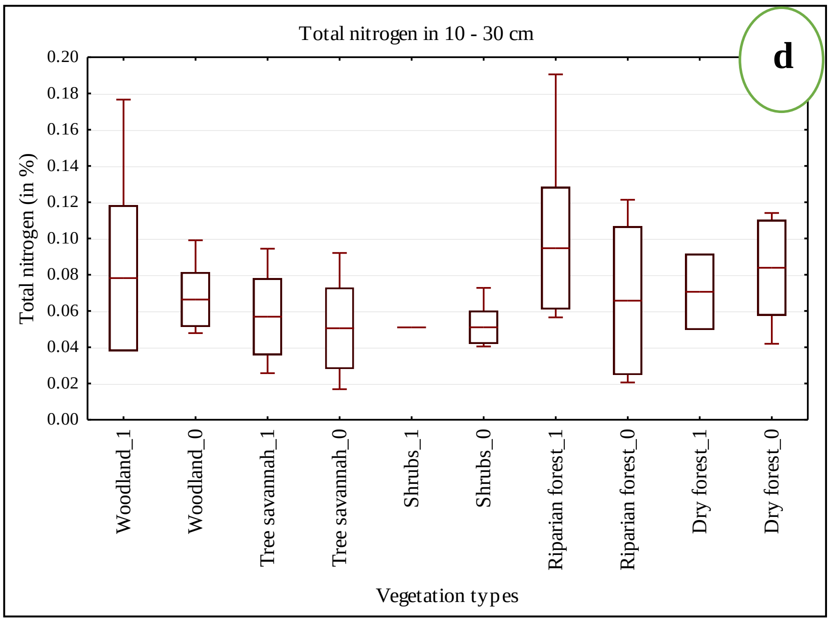
A contrasting analysis for protected - unprotected areas showed that protection status had significant effect on the SOC amounts (Figures 3.4a and b). Regardless of LUC types, mean topsoil SOC was higher than subsoil SOC, except the riparian forests in UPA. Dry forests of both PA and UPA, and protected riparian forests showed highest values (over 3 %) of topsoil SOC. In the topsoil layer of PA, the rank order was riparian forests > dry forests > woodlands > tree savannahs whereas the land cover types ranked as dry forests > woodlands > shrubs > tree savannahs > riparian forests in UPA. In the subsoil, SOC content ranked in the following order: woodlands > riparian forests > dry forests > tree savannahs within PA. Meanwhile the rank in UPA was riparian forests > dry forests > woodlands > shrubs > tree savannahs.

Similar to the trends of SOC, TN exhibited high variability in the topsoil layer in dry forests and riparian forests (Figures 3.4c and 3.4d). TN content was low in the lower 20 cm depth (< 0.10 %) for all LUC types. Meanwhile, riparian forests of PA (> 0.20 %) and dry forests (> 0.15 %) showed the highest TN values in the upper 10 cm depth, and decreasing trends of riparian forests > dry forests > woodlands > tree savannahs, and dry forests > woodlands > riparian forests > shrubs > tree savannahs, for PA and UPA, respectively. The corresponding trends of TN in the subsoil were riparian forests > woodlands > tree savannahs > dry forests and dry forests > woodlands > tree savannahs > riparian forests > shrubs.

The trends in LUC types for TN and SOC at both depths indicate forests and woodlands to store more of these parameters regardless of the protection status. The implication is that these LUC types possess high potential for SOC and TN storage and resilience capacities depending on their management regime. It can also be concluded high amounts of SOC and TN in both PA and UPA are associated with good land cover.





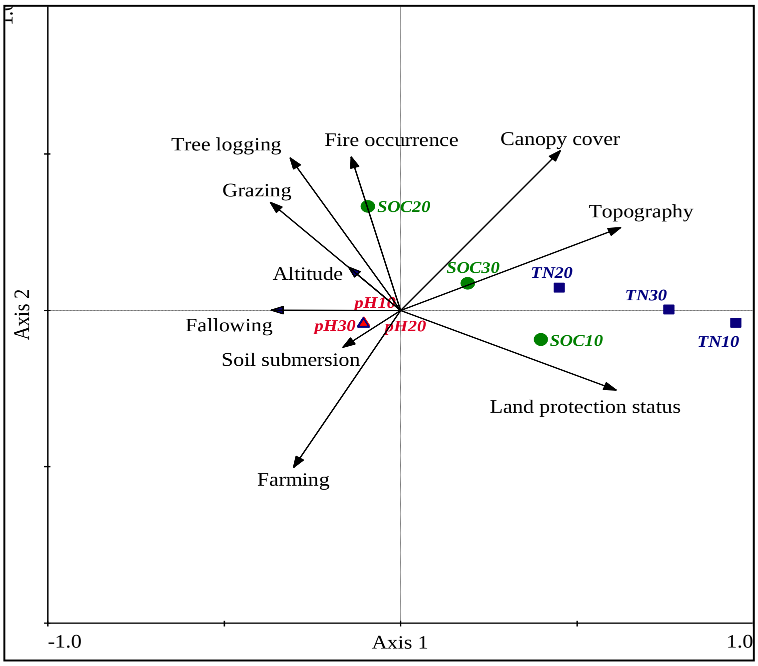


***Note**: The Boolean codes (0/1) mean unprotected (0) and protected (1) lands.*

Figure 3.4 Effects of protection status on SOC (a and b) and TN (c and d)

3.3.4. Interactions between soil properties and environmental variables

The outputs from CCA (Figure 3.5) revealed the relationships between measured pH, SOC and TN, and the site ecological variables. In the CCA, the first three explanatory variables affecting mostly SOC and TN in the top 10 cm, were topography, vegetation canopy cover (related to LUC types), and land protection status. SOC in the top 10 cm and the overall 30 cm was often affected by land protection status, topography and canopy cover. However, subsoil SOC tend to be higher when fire, grazing and tree logging occurred in the sites. This suggests that the occurrence of fire and tree logging often reduce the availability of litter and destroy soil micro-fauna that decomposes litter into SOC. Since grazing occurs in bush fallows where fire occurrence is high, it is evident that fallowing has the same effect as fire and logging on SOC in soil. Though pH is less affected by *in situ* conditions, fallowed soils and waterlogged soil conditions are related to high pH values. It is shown that human-related activities are negatively correlated with high canopy cover, high topography and protected lands, which positively increase soil nutrient contents.



Note*: Green dots represent SOC, red triangles are the pH values, and the blue squares denote TN for the upper 10 cm, lower 20 cm and the overall 30 cm, respectively.*

Figure 3.5. Relationships between ecological variables, SOC, TN and pH

Furthermore, both negative and positive correlations were observed between topographic indices and environmental parameters, and SOC and TN (Table 3.3). Strong positive correlations ($p \leq 0.05$) were found between soil chemical properties at both depths, except SOC in the 20 cm depth, which exhibited positive but not significant correlations with SOC in the 20 cm, and TN in 10 cm depths. Except subsoil SOC, pH had slight negative effects on TN and SOC at the two depths. With regard to the ecological conditions, topsoil organic carbon was positively correlated with TPI whereas subsoil TN was strongly affected by slope. C and K factors had negative effects on SOC and TN at both depths. R factor had similar negative effects only in the upper 10 cm. These negative correlations between soil erosion factors and soil nutrients indicated that soil loss negatively affects soil nutrient availability.

Additionally, factor analyses were conducted to extract the main factors controlling the distribution of SOC and TN (Table 3.4). The varimax indicated that the four first factors explained about 82 % of the total variance. Factor 1 showed high loadings for SOC and TN at all depths, indicating the factor of soil fertility. In factor 2, SPI, TWI, and CA exhibited positive high loadings (0.70 - 0.89), indicating the direct influence of soil moisture and drainage on soil properties. This factor defines mostly topographical conditions likely to affect the soil properties. Similarly, slope, altitudes, CA and SPI defining the factor 3 indicated the influence of landscape positions and soil susceptibility to erosion on soil properties. Higher topographic indices were often negatively correlated with higher soil vulnerability to erosion, resulting in negative effects on soil chemicals. The fourth factor is less informative though it indicated high negative loading value of - 0.74 for subsoil SOC.

Table 3.3 Correlation matrix between soil chemical properties, different pedologic and topographic parameters

	SOC10	TN10	SOC20	TN20	SOC30	TN30	pH10	pH20	C-factor	K-factor	R-factor	SPI	TWI	Alt	Alt.ch	Slope	CA
TN10	0.7704*																
SOC20	0.1174	0.1408															
TN20	0.6408*	0.6575*	0.2805*														
SOC30	0.8718*	0.6964*	0.5889*	0.6599*													
TN30	0.7903*	0.9667*	0.1999	0.8285*	0.7419*												
pH10	-0.0107	-0.028	0.0107	-0.1003	-0.0035	-0.0549											
pH20	-0.0198	-0.0867	0.0618	-0.0281	0.0144	-0.074	0.1007										
C-factor	-0.1141	-0.2151	-0.2873*	-0.2546*	-0.2346*	-0.2464*	0.0268	-0.1428									
K-factor	-0.1483	-0.0963	-0.1848	-0.1273	-0.2118	-0.1148	0.0815	-0.0318	0.4415*								
R-factor	-0.0173	-0.0104	0.0045	0.0091	-0.0118	-0.0046	0.012	0.0274	0.3316*	0.5170*							
SPI	-0.0071	0.0932	0.1901	-0.1453	0.088	0.0199	0.1099	-0.0881	-0.1152	-0.0136	0.0138						
TWI	0.0017	0.1093	0.1098	-0.0628	0.0555	0.0599	-0.1121	-0.1281	-0.0482	-0.0044	0.053	0.5546*					
Alt	0.01	0.0325	0.1004	0.1546	0.0577	0.0767	-0.2712*	-0.0217	-0.3173*	-0.3656*	-0.033	-0.1249	-0.1935				
Alt.ch	0.0044	0.0223	-0.0338	0.1155	-0.0131	0.0558	-0.1827	0.1098	-0.2340*	-0.122	-0.1139	-0.0927	-0.3036*	0.5413*			
Slope	0.2222	0.0698	0.0489	0.2614*	0.205	0.1407	-0.1297	0.0339	-0.2297*	-0.0537	0.0266	-0.0666	-0.3704*	0.4671*	0.4499*		
CA	-0.0079	0.0923	0.1902	-0.1463	0.0874	0.0189	0.1116	-0.0873	-0.1149	-0.0135	0.0137	1.0000*	0.5539*	-0.1257	-0.0928	-0.068	
TPI	-0.2597*	-0.1758	0.0499	-0.1974	-0.1867	-0.1977	0.2111	-0.0175	0.0861	0.019	-0.0534	-0.0483	-0.4137*	-0.0044	0.2537*	-0.0582	-0.0465

***Note**: Correlation coefficients are displayed with star at 95 % CI. SOC10, SOC20 and SOC30 represent SOC for the 0 - 10 cm, 10 - 30 cm and the 0 - 30 cm. Ditto for TN and pH.* K, R, LS *and* C *factors are the input parameters of the RUSLE model.* SPI = *stream power index;* TWI = *topographic wetness index,* Alt.ch = *altitude above channel;* D_road = *distance to the main road;* D_village = *distance to a village center;* Land_man = *Land management regime;* GSL = *gross soil loss.*

Table 3.4. Varimax of loading factors of the principal factor analysis

Variables	Factor 1	Factor 2	Factor 3	Factor 4
SOC10	**0.869**	-0.015	-0.246	0.094
TN10	**0.873**	0.095	-0.218	0.183
TN20	**0.837**	-0.198	-0.110	0.014
SOC30	**0.903**	0.086	-0.031	-0.286
TN30	**0.933**	0.004	-0.199	0.140
SOC20	**0.398**	0.198	**0.343**	**-0.736**
Alt	0.182	**-0.390**	**0.553**	0.221
Alt.ch	0.102	**-0.406**	**0.477**	0.263
Slope	0.262	**-0.365**	**0.355**	0.260
CA	0.063	**0.887**	**0.355**	0.191
SPI	0.064	**0.887**	**0.355**	0.193
TWI	0.059	**0.697**	-0.024	0.088
K-factor	-0.249	0.099	**-0.489**	0.132
R-factor	-0.066	0.079	**-0.326**	0.097
C-factor	**-0.340**	0.041	**-0.504**	0.059
pH10	-0.082	0.132	-0.116	-0.198
pH20	-0.023	-0.107	0.0701	-0.125
TPI	-0.238	-0.180	0.145	-0.180
Eigen values	4.43	2.69	1.83	1.07
% of variance explained	36.07	21.87	14.90	8.70
Cumulative % variance explained over factors	36.07	57.94	72.84	81.54

Note: SOC10, SOC20 and SOC30 stand for SOC in the topsoil, subsoil and the overall 30 cm. Ditto for TN and pH. K, R, LS *and* C *factors are the RUSLE input parameters.* SPI = *stream power index;* TWI = *topographic wetness index,* Alt.ch = *altitude above channel;* D_road = *distance to the main road;* D_village = *distance to a village center;* Land_man = *Land management regime;* GSL = *gross soil loss.*

3.3.5. Soil chemical properties-environment interactions in agro-systems

Figure 3.6 obtained from a CCA revealed that the farm and fallowed lands are quite similar according to soil characteristics (clustered red triangles). This is because fallowed lands are often less than three years, i.e. not yet recovered. However, the main factors that differentiated sample sites are the ecological variables (black dots). In the right half of the axis 1 of CCA, fire and grazing are the predominant environmental disturbances occurring in fallows (green rectangles). Fallows in the Mo basin are bush fallows where cattle breeders used to set fire for forage. Axis 1 highlighted that chemical properties in farmlands are somehow slightly higher than fallows, probably due to fact that farms are still fertile and have not reached a critical fertility level to be set into fallows. Tree logging related to canopy cover occurred in one site due to the recovering of the vegetation. These environmental disturbances did not occur in farmlands (red diamonds) where tree canopy cover is low, and fire occurrence and grazing are inexistent. Among other environmental features, topography, altitude and woody species richness do not differentiate significantly farms and fallows.

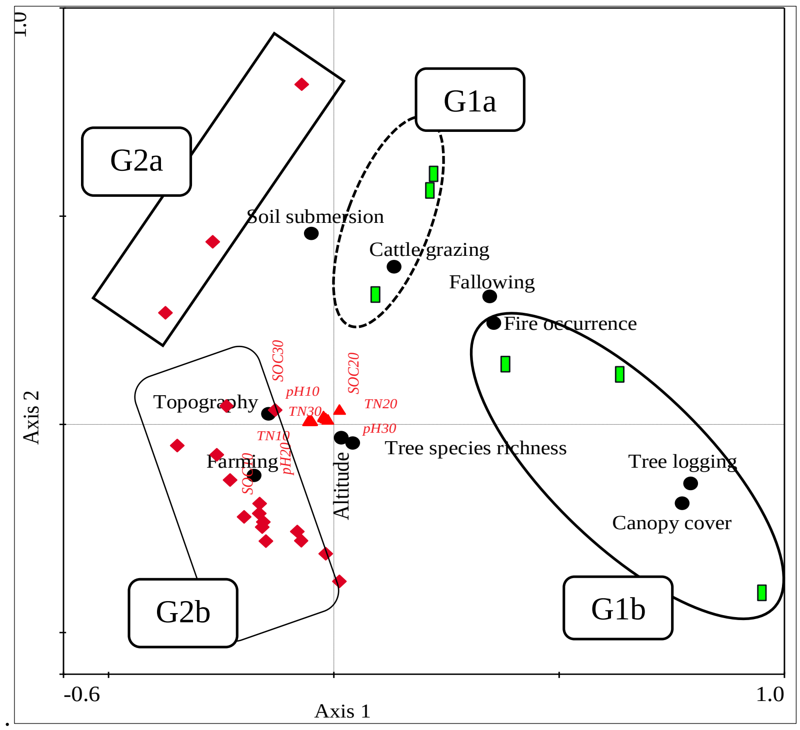


Figure 3.6. CCA of the distribution of soil samples and properties in agro-systems

Further analysis in discriminating sample sites was through the dendrogram (Figure 3.7). It provided a full discrimination among soil samples that were collected in agro-systems according to the level of their similarity/dissimilarity shaped by the intrinsic prevailing environmental variables. At 0.6 level of similarity (dashed red line), two groups of agro-systems are discriminated. On one hand, G1 composed of 6 sites corresponds to the fallows or abandoned lands after years of cropping. G1 comprises fallows located at any topographical position and experiencing activities such as fuel wood gathering, grazing and fire occurrence. On the other hand, G2 is a cluster of 18 sites located in current farmlands.

In these sites, no fire, logging, and grazing activities occur. At 0.8 level of dissimilarity (black solid line), each of the aforementioned groups is subdivided into 2 sub-groups. G1 clustered two subgroups G1a and G1b. G1a is characterized by sites located on riverbanks (S4 and S8) and flat terrain (S34). Woody plant species richness ranges from 15 to 22 with slight vegetation recovering (high canopy cover). In contrast, sites of G1b are located on waterlogged soils and have low species richness (6-15). They also experienced grazing and fire effects. On the other side, G2 subdivisions highlighted the discrimination of G2b corresponding to a mosaic of farm sites located mostly on waterlogged soils (except S33 and S43) at variable topographic positions. G2a (S35, S22, and S13) were located on waterlogged soils where grazing activities occurred. A further similarity analysis at 0.9 (black dash-dot line) showed the same subgroups for G1a, G1b, and G2a. However, G2b discriminated G2b2 (S33 and S43) composed of farms located on waterlogged soils on top-slope. G2b1 corresponds to high land farms without waterlogging. Grazing, fire and tree logging are quite inexistent in these sites.

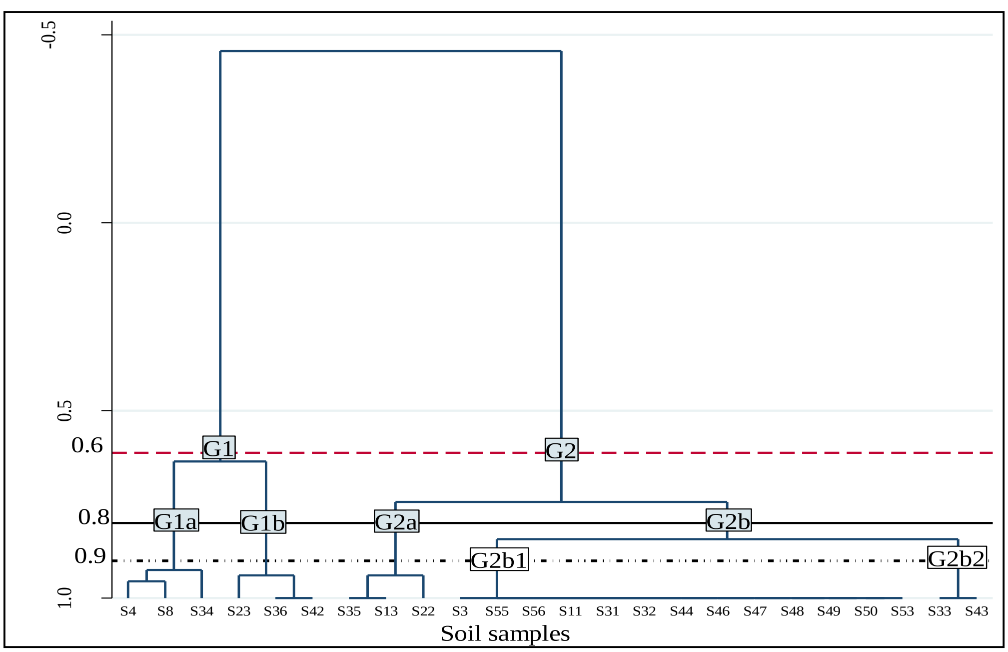


Figure 3.7 Similarity/dissimilarity amongst sample sites in agro-systems

3.3.6. Spatial distribution of TN and SOC in the Mo basin

Figure 3.8 displays the general distribution of TN and SOC stocks at all depths according to the spatial patterns of LUC in the year 2014.

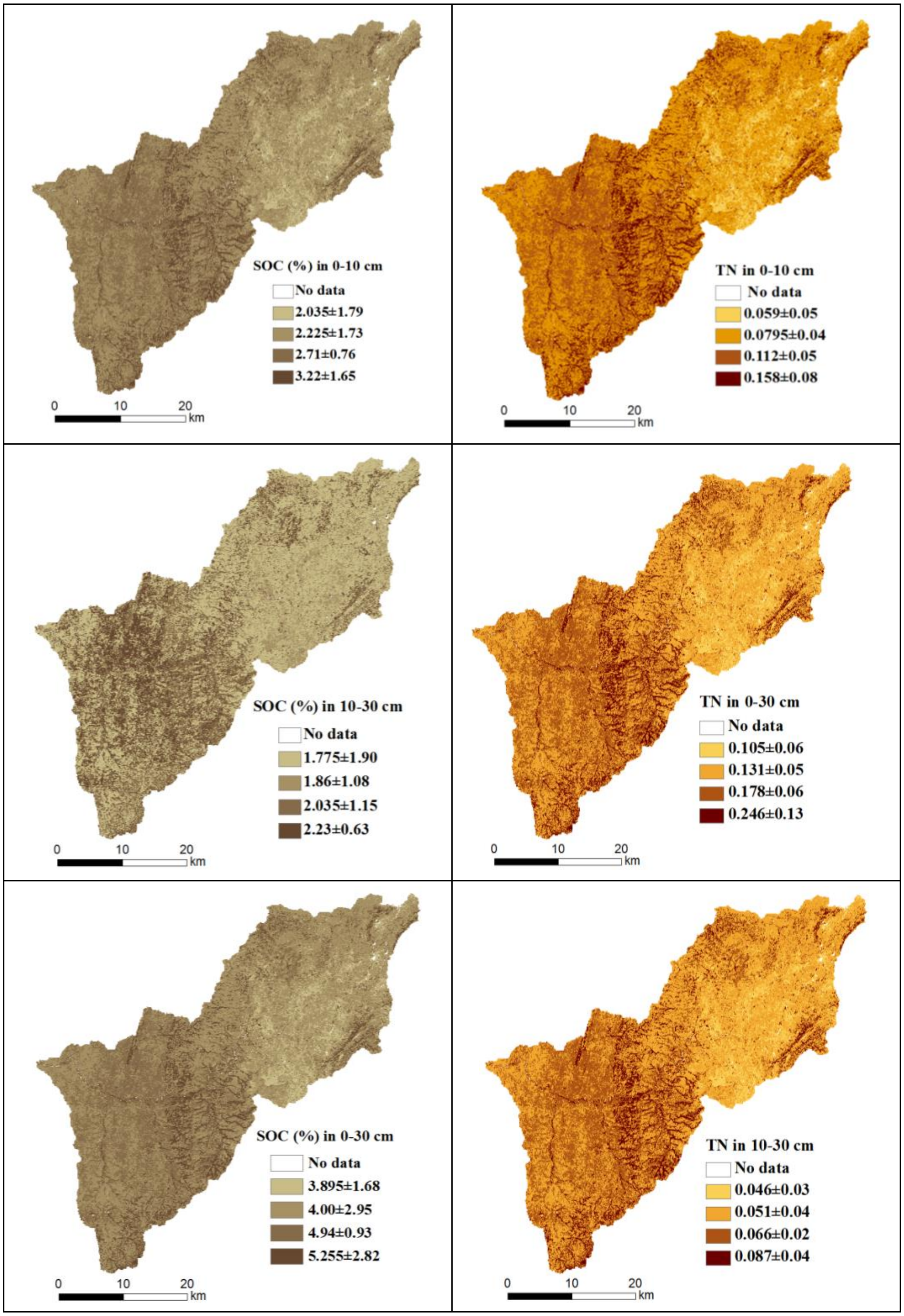


Figure 3.8. Spatial patterns of SOC and TN for the Mo River basin

As indicated in Figure 3.8, the distribution of soil nutrients showed similar spatial patterns for both TN and SOC at all depths except the subsoil SOC. At this latter depth, woodland exhibited highest SOC. As the vegetation cover increased from croplands (treeless lands) to riparian and dry forests (closer lands), the amounts of soil nutrients also increased. Spatially, PA had soils with high SOC and TN contents, especially inland valleys, streamsides, and low slopes. However, the northern parts of the basin exhibit patterns of high SOC and TN contents, suggesting a good positive relationship between vegetation cover and those soil properties.

3.4. Discussion

3.4.1. Patterns of SOC and TN storage in Mo river basin

The analysis of soil data within the Mo basin showed a spatial variability of SOC and TN in relation to LUC types, land protection status and landforms. In the first instance, the influence of LUC types on SOC and TN was evident through their high contents in natural lands compared to agricultural lands (up to a difference of 2.2 %). Similar observations have been reported in the northern landscapes of Togo (Sebastia et al., 2008) and south-eastern landscapes of Ethiopia (Abera & Belachew, 2011). Furthermore, other findings in the same study region reported similar SOC-richer dry forests than savannahs (Fontodji et al., 2009). In this study, SOC in 0 – 10 cm depth (2.04 – 3.22 %) for all LUC types is slightly higher than those reported by Fontodji et al. (2009) for a topsoil of 0 – 20 cm depth (1.5 – 3.15 %). This ascertained the fact that the deeper the soil layer, the lower its contents in SOC and TN. In line with Bessah et al. (2016), the highest SOC contents was recorded in the topsoil of all LUC types but varied across them because land use/management and cover might have significant influence.

On the basis that soils of natural vegetation provided the baseline for the potential fertility, similar differences in terms of SOC and TN contents between the croplands and forest soils have been noted in northern Togo, as a consequence of agricultural land use inducing the loss of soil fertility (Poch and Ubalde, 2006; Sebastia et al., 2008). The cultivation processes and other practices inducing the loss of vegetation cover caused a significant reduction of SOC and TN inputs consecutive to the loss of native vegetation. In addition, Gidena (2016) reported that the low SOC and TN in agricultural lands might be due to the effects of tillage that induces the loss of C as CO_2 by breaking up soil aggregates and exposing the OM to microbes. With regard to TN, the correlation outputs showed that high contents are strongly correlated with high SOC, indicating that practices inducing SOC depletion (Emiru & Gebrekidan, 2013; Xue et al., 2013) would decreased TN contents while conservation would lead to consecutive accumulation of chemicals.

Furthermore, this study conducted in mountainous areas showed that geomorphic positions strongly determined the spatial distribution of SOC and TN, with richer soils in lower topographic positions. Ofori et al. (2013) made similar observations and related that to the surface runoff which increases the nutrient concentration along the toposequence by carrying them downward slopes, especially in the topsoil. Similar to the effects of LUC types, topography induced spatial variability in SOC and TN with topsoil richer than subsoil. However, the spatial patterns of soil conditions, especially its nutrient contents may be induced by many other factors.

3.4.2. Factors controlling the distribution of SOC and TN in Mo river basin

Beside topography and LUC types, other factors such as *in situ* ecological variables and human disturbances affect the spatial distribution of the soil nutrients (Meng et al., 2014;

Wang et al., 2013b; Wang et al., 2010). This study revealed that land use/cover types, as reported by Yao et al. (2010) and Bessah et al. (2016), affect SOC and TN availability. In the mid, Fontodji et al. (2009) showed that charcoal production reduces the OM availability at the kiln sites, indicating that tree cutting and fire adversely affect soil stability and functions of carbon and nitrogen reservoirs (Novara et al. 2012). As a result of high SOC and TN content in low lands, it is reported that overland flow and runoff play an important role in the nutrient transport and sedimentation, increasing the SOC and TN stocks in inland valleys, lowlands and riverbanks (Liu & Bliss, 2003; Yadav & Malanson, 2013). As indicators of surface runoff and soil erosion effects, the negative correlation between the RUSLE factors (K, C and R factors) and the SOC and TN, especially in the topsoil, indicated that erosive rainfall on high erodible and less covered soils deplete the topsoil OM and nitrogen. These effects were less significant in the subsoil, confirming the high sensitivity of the topsoil to management and erosion in the landscape. Though it is established that high amounts of rainfall induces high biomass production and low OM decomposition in soils (Wiesmeier et al., 2014b; Wiesmeier et al., 2013a), its intensity could have adverse effects on nutrient storage, due to rapid surface runoff that causes the detachment and leaching of finer sediment (Cheng et al., 2010).

3.4.3. Land use/management inducing variability in the SOC and TN

As it is common, current farming systems in the Mo basin rely on the natural land productivity for crop production. This suggests that they induce loss of fertility after years of farming through a depletion of SOC, TN and other nutrients in relation with the management systems. Compared to natural lands, cultivated lands exhibited low records of SOC (3.9 %) and TN (0.11 %) for a 30 cm depth, but they still have soil high quality thresholds for agricultural purposes, confirming that traditional tillage methods do not excessively disturb soil layers to a depth greater than 30 cm depth as reported by (Vagen & Winowiecki, 2013). In addition, this study revealed high SOC and TN in farmlands compared to fallows probably because the former are still fertile and have not reached a critical impoverishment level to be turned into fallow. This finding contrasted the results of Novara et al. (2014) who found that land abandonment after many years increased the soils nutrients. This may be due to the age of the fallows in this study, which are abandoned lands of less than 3 years, suggesting that replenishment of SOC and TN in soils after years of cropping does not occur immediately after land abandonment.

Furthermore, it has been shown that land clearing and continuous cultivation reduced more than 50 %, even worse, of SOC and TN compared to undisturbed native soils (Knops & Tilman, 2000; Parras-Alcaantara et al., 2013; Solomon et al., 2000). In this study, PA soils stored more TN and SOC at the two depths than human-affected soils. This highlighted the positive effects of land protection on healthy soil, appealing for necessary and important measures to enhance land conservation and carbon and nitrogen stocks for both food security and climate change mitigation.

3.5. Conclusion and implications for sustainable land management and conservation

This chapter evaluated the soil conditions in the Mo River basin through an analysis of SOC and TN contents in the various vegetation types and topographical positions in relation with the ecological variables controlling their spatial distribution. All sites exhibited decreasing TN and SOC with increasing soil depth indicating that the topsoil concentrated more SOC and TN. Measured SOC and TN contents for 0 – 10 cm, 10 – 30 cm and 0 – 30 cm depths varied both within and between sites. In the topsoil (0 – 10 cm), the average TN varied from 0.056 % in fallows to 0.159 % in dry forests whereas in the subsoil (10 – 30 cm), the lowest (0.043 %) and the highest (0.091 %) average TN contents occurred in the woody savannahs and dry forests, respectively. As far as SOC in the topsoil is concerned, a significant difference was observed between vegetation types, with an average SOC ranging from 1.81 % in fallows to 3.58 % in dry forests. Dry forests, gallery forests, and woodlands were significantly SOC-richer. In contrast, SOC in subsoil layer did not show any significant difference in relation with the vegetation types. Generally, SOC and TN are highly concentrated in the soils under healthy vegetation whereas they decreased with increasing soil depths. This study further showed that the SOC density exhibited high spatial variability in relation to the terrain variability. In the topsoil, flat terrain (1.89 %) and riverbank (2.16 %) were less SOC-richer whereas lower (2.63 %) and mid-slopes (3.53 %) exhibited the highest SOC. River banks (2.06 %) and summits (2.03 %) recorded the highest average SOC. With regard to the TN, its average content in the topsoil varied significantly from 0.05 % on flat terrain to 0.150 % on mid-slopes whereas the lowest (0.046 %) and highest (0.084 %) averages were observed on flat terrain and mid-slope, respectively. Generally, environmental disturbances such as fire, grazing, soil erosion and land conservation status are the major factors influencing the contents and spatial variability of SOC and TN. This study achieved the diagnostic of SOC and TN as key characteristics measuring threats to soil health, and identified the drivers of soil-based ESS provision for climate mitigation and food security.

CHAPTER 4: PATTERNS OF VEGETATION IN RELATION TO BIOPHYSICAL AND HUMAN FACTORS IN THE MO BASIN [2]

4.1. Introduction

In tropical regions, natural landscapes provide many functions and services, such as biodiversity conservation, climate regulation and livelihood support to millions of people (Shackleton et al., 2007; Tindan, 2015). Most populations depend on these natural resources, which they manage and conserve according to their usage, and available resources (Ouedraogo et al., 2013; Pouliot et al., 2012; Steele et al., 2015). This situation underscore the institution of management options comprising protected areas (PA) and community forest zones, to maintain ecosystem integrity. Unfortunately, increasing human pressures on land resources have affected these PA to negate their biological conservation role (Damnyag et al., 2013; Dimobe et al., 2014; Folega et al., 2014b; Wala et al., 2012). Human disturbances therefore appear to control natural ecological factors in shaping and changing the functions and structure of the landscapes.

In Togo, especially in the mountainous ecosystems of Central region, numerous human pressures are affecting the landscape patterns, even in PA. In those areas, intensive wood extraction of timber, firewood and charcoal production, and small scale farming system significantly affect the landscape structure and induce degradation of natural ecosystems (Dourma et al., 2009; Wala et al., 2012). Understanding the interactions between human imprints and biophysical components defining the landscape heterogeneity is therefore fundamental for landscape management and biological conservation (Ali et al.,

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2014). In addition, the spatial information of the landscape patterns are required to assess the extent of degradation and to guide the development of sustainable management options.

Situated in one of the richest landscapes in Togo, the Mo River basin is undergoing continuous transformation. Despite its importance, the ecological status and appropriate conservation management remain poorly understood. Though PA are demarcated in the region to ensure biological conservation, the supporting public policies have failed due to weaknesses in law enforcement and illegal incursions (Wala et al., 2012). Attempts to propose pathways for sustainable management of landscapes were undertaken fundamentally on the characterisation of vegetation structure and floristic composition in relation to environmental variables and human disturbances (Dourma, 2008; Woegan, 2007). Not much attention has been paid to the spatial patterns of the evaluated land resources their potential availability. Furthermore, the soil conditions in these ecosystems are not well resourced despite the correlation between soil conditions and vegetation patterns (Galal & Fahmy, 2012). This gap informed the approach used in this study that integrates geographical information systems (GIS) and remote sensing (RS) with field measurements to produce spatially explicit landscape patterns and its indices of degradation in order to facilitate the sustainable management of the multifunctional landscapes of the Mo River basin. Specifically, the study aimed at determining the vegetation patterns in relation to biophysical and human factors prevailing in the landscapes of the Mo River basin. The underlying hypothesis was that PA exhibit better indicators of land conservation performance in terms of the stand structure and characteristics as well as edaphic-ecological variables.

4.2. Material and Methods

4.2.1. Study area

a) Vegetation and soils in the Mo basin

The Mo basin is part of the Soudan-Guinean zone located in the ecological zone 2 of Togo. The vegetation is characterized by a mosaic of dry and riparian forests, woodlands, savannahs and agro-systems (Woegan, 2007). The prominent plant species in dry forests include *Isoberlinia doka*, *Isoberlinia tomentosa*, *Monotes kerstingii*, *Detarium microcarpum* and *Uapaca togoensis* (Dourma, 2008; Dourma et al., 2009; Woegan, 2007). Species as *Berlinia grandiflora*, *Afzelia Africana*, *Ceiba pentandra* and *Khaya senegalensis* are dominantly observed along riversides in riparian forests. Savannahs and shrubs are made up of woody species and grasses dominated by Andropogonae and Hyparrhenia.

The major parent material of the soils of the Mo River basin is the sericite and muscovite dominant quartzites. A detailed soil information is lacking at the local level of the Mo basin. However, Lithosols and ferruginous tropical soils are the dominant soil types with some patches of ferralitic soils (Lamouroux, 1969). Soils are less developed and shallow with high proportion of sand, coarser elements and rocks.

b) Network of protected areas

The study area encompasses three protected areas (PA): Fazao-Malfakassa National Park (FMNP), Aledjo Wildlife Reserve (AWR) and Kemeni Forest Reserve (KFR). FMNP covers about 192,000 ha between 8° 20’N and 9° 30’N and 0° 35’N and 1° 02’E. This PA experiences human disturbances, especially at the edges where riparian populations often collect forest-based resources. Between 1978 and 2011, the PA has experienced a loss of 22,073 ha i.e. 10 % of its initial area (Aboudou, 2012). The AWR, located in the area of

Aledjo, between 9°11 and 9°17 N and 1° and 1°24 E, covers about 785 ha in the ecological zone 2 of Togo. KFR is adjacent (eastern side) to AWR and is of minor importance in the national network of PA. Regarding the AWR and KFR, the prevalent human activities include fuelwood and NTFPs collection (Wala et al., 2012; Woegan, 2007).

c) Fauna

FMNP is rich in megafauna composed of *Hippotragus equinus*, *Kobus spp.*, *Tragelaphus scriptus*, *Limnotragus spekei*, *Cephalophus spp.*, *Sylvicarpra spp.*, *Hylochoerus meinertzhageni*, *Loxodonta africana*, *Papio anubis*, *Erythrocebus patas*, *Cercopithecus sabacus*, etc. In addition to other mammals (primates), a high diversity of reptiles (turtles, crocodiles, snakes), birds and micro-fauna is common in the park. AWR and KFR are relatively poor in megafauna due to the high topography and their narrowness. However, some primates, reptiles, birds, and micro-fauna are observed in these PA (MERF, 2002). Ichthyologic fauna is diversely observed in the rivers and ponds of the basin.

4.2.2. Deriving landscape patterns based on contemporary LUC for Mo basin

In order to provide a spatial view of the different vegetation types in both PA and UPA of the Mo River basin, this study relied on Landsat 8 image downloaded from https://earthexplorer.usgs.gov. The full procedure for deriving the contemporary (2014) spatial patterns of vegetation is presented in Chapter 5 dedicated to LUC mapping.

4.2.3. Collection of vegetation data and ecological variables

Topography was a major constraint during the sampling. Consequently, 75 forest inventory plots were set along the catena without any predefined plot number for each location. The plots were randomly set according to accessibility, the representativeness of flora biodiversity and the vegetation homogeneity in such a way to represent the different vegetation types in the landscape. In total, the number of plots varied according to protection regime (39 and 36 plots in protected and unprotected areas, respectively) and vegetation types (9 in dry forests, 19 in riparian forests, 10 in shrubs, 19 in tree savannahs, and 18 in woodlands). Squared plots of 20 m x 20 m and 30 m x 30 m were set in dry forests and savannahs, respectively. Plot dimensions were 50 x 10 m in riparian forests in order to match the linear shape and the width of the ecosystems in the savannah-dominated landscapes (Dimobe et al., 2014; Folega et al., 2010; Wala et al., 2012). In each plot, the following attributes of woody species were recorded for each tree: species name, total height, diameter at breast height (DBH), crown diameters (North-South and West-East directions). These were georeferenced using a GPS Garmin 62S. Human disturbances, i.e., fire occurrence, grazing, selective tree logging and charcoal production, were recorded as presence/absence (1 = presence, 0 = absence). Other ecological attributes such as vegetation type, soil type, soil waterlogging or soil submersion (0 = No, 1= Yes), canopy cover density (coded as 0 = very low, 1 = low, 2 = medium, and 3 = high) and protection status (1 = protected/0= unprotected) were also collected. For data processing purpose, six major topographical positions were selected (See Chapter 3 for details).

In addition to the aforementioned ecological characteristics, soil data were used to provide soil chemical properties of each forest inventory plot. The detailed procedure of soil sample collection and management is provided in Chapter 3.

To assess the potential influence of topography on vegetation types in Mo basin, some topographic-based indices were derived from the SRTM-DEM. These indices included terrain attributes such as slope, Topographic Wetness Index (TWI), and mean altitude above the channel level (Alt.ch). The procedure for computation is described in

Chapter 3 Section 3.2.4. The TWI is a topographic variable indicating the spatial distribution of soil moisture conditions, a potential indicator of species preference to moisture. Alt.ch is used to indicate the potential depth to groundwater/free water for plant species, and therefore to moisture content in the substratum. Maps of slope, TWI and Alt.ch were derived using SAGA GIS 2.0.8 platform that embeds the algorithms for computing these variables. Soil texture was obtained from the Harmonized World Soil Database (HWSD) but was not considered in detail analyses since no difference existed in the soil properties according to soil types. The maps were exported to a GIS for data extraction to the GPS of the plots.

4.2.4. Vegetation and ecological data analyses

The ordination method was used to identify environmental gradients defining species distribution and landscape patterns. Among the various multivariate methods used for this purpose in plant community analyses, the indirect gradient algorithms of Detrended Canonical Correspondence Analysis (DCA) was used to analyse the variation of plant communities and their relationships with environmental variables. The first reason is that they are commonly used for plant community analyses (Dimobe et al., 2014; Folega et al., 2014a; Kebede et al., 2013; Tavili & Jafari, 2009; Wale et al., 2012; Zhang & Zhang, 2010). The second reason is that the dataset was too heterogeneous and too many species deviated from the assumed model of linear response (Leps & Smilauer, 2003). In this sense, the obtained gradients length during trials, which measures the beta diversity in community composition i.e. the extent of species turnover along the individual independent ordination axes, exhibited values between 3 and 4. In such cases, it was compelling to select unimodal method using the indirect detrended DCA method by segments down-weighting rare species without any transformation of the initial information. DCA methods summarize variation in the relative frequencies of the response variables (species). An important implication is that these methods cannot work with ‘empty’ samples, i.e. records in which no species is present (Leps & Smilauer, 2003). The DCA were performed in Canoco 4.5 and CanoDraw 4.1 for Windows. Two wys species indicator (TWINSPAN) in Community Analysis Package (CAP 2.15) clustered plant communities according to their level of similarity. First, a matrix of 75 plots x 142 plant species subjected to a DCA ordination did not help in depicting plant communities along any gradient. Therefore, the data were split into 36 unprotected relevés and 39 sampling plots in PA. The two matrices, 39 sampling plots x 121 species and 36 relevés x 100 species were separately subject to ordination, respectively for PA and UPA.

Three measures of species diversity, i.e. species richness (*S*), Shannon-Wiener’s species diversity index (*H’*) (Equation 4.1.), and Pielou equitability index (*E*) (Equation 4.2), were computed to characterise each plant community whereas Jaccards’s index was used to assess the level of similarity among the plant communities. Species richness (*S*) was computed for each plant community as the total number of woody species recorded in the relevés of the community. Because of different plot sizes, mean species richness at plot level was calculated as the number of species in a plot divided by the log of the area sampled (White et al., 2014).

Shannon-Wiener’s species diversity index (*H’*) is calculated using equation 4.1:

$$H' = -\sum_{i=1}^{S}(Pi)log_2(Pi) \qquad \text{(Equation 4.1)}$$

where *Pi* = *Ni/N* with *Ni* = the number of individuals of species *i* and *N* = the total number of individual of all recorded species.

Pielou equitability index (*E*) is calculated based on *H'* and *S* as:

$$E = -\frac{H'}{log_2 (S)} \qquad \text{(Equation 4.2)}$$

Equation 4.3 is the formulae for computing the plot basal area at breast height (G in m^2 ha^{-1}, sum of cross-sectional area of all trees with DBH ≥ 10 cm in a given plot).

$$G = \frac{\pi}{4A} \sum_{i=1}^{n} 0.0001\sqrt{di} \qquad \text{(Equation 4.3)}$$

where n is the number of trees recorded in the plot, A is the sampled plot area (plot size, in ha), and di is the DBH (in cm) of the i^{th} tree.

In addition, vegetation stands were characterized by computing their tree densities. Regeneration of all mixed species was also calculated to provide an overview landscape dynamics. Average soil chemical properties (pH, SOC, and TN), and stand conditions (mean indices of soil waterlogging, fire occurrence, canopy cover, selective tree logging, and Alt.ch) were calculated for the various vegetation types in both PA and UPA.

Similarity between plant communities was assessed using Jaccards's index (*Sij*) and plotted using TWINSPAN in CAP 2.15. The performance of Jaccards's Similarity was used to compare the plant communities under the different protection status.

$$Sij = \frac{C}{A+B-C} \qquad \text{(Equation 4.4)}$$

where A is the number of species belonging to the plant community i , B is the number of species belonging to the plant community j , and C the number of species belonging to both plant communities i and j. If $Sij \geq 50$ %, communities exhibit similarity; otherwise (i.e. $Sij \leq 50$ %), there is no similarity (Woegan, 2007).

Supplementary statistical analyses were performed through analysis of variance to compare characteristics of the different plant communities, assuming equality of variance and normality in data distribution. Outputs from analyses of variance were contrasted using post-hoc multiple comparison of Tukey at $p < 0.05$. Pairwise correlation was performed to depict the relationship between biophysical and ecological variables within the plant. A 3-parameter Weibull function was used to give a pictorial view of the distribution of the tree heights and diameters in the different vegetation types.

4.3. Results

4.3.1. Contemporary landscape patterns from satellite image

The aerial distribution and spatial patterns of the contemporary vegetation patterns are fully detailed in Chapter 5 Section 5.3.1. In general, the southern and southwestern parts of Mo basin present wilder landscapes attributable to protection status and the increasing distance from settlements. However, these factors in combination with biophysical and soil conditions could provide explanations for the landscape patterns.

4.3.2. Discrimination of vegetation types in Mo river landscapes

The relation between human disturbances, land protection and biophysical factors in the 75 plots showed that the first two DCA axes (Figure 4.1) carried out 33.2 % of the cumulative variance of species-environment interactions (Appendix 2). Axis 1 denoted a negative gradient of human disturbances in opposition to positive correlation with topography and protection status. Axis 1 of the DCA correlated positively with protection status (0.58) and topography (0.63) but negatively correlated to human footprints (-0.53 and - 0.63 for tree logging and fire occurrence, respectively). Meanwhile, axis 2 showed a positive correlation with protection status (0.64). These results indicate that the most important ecological factors determining landscape patterns are protection status and topography, in contrast to human disturbances. However, the ordination plot did not allow a net discrimination of the different vegetation types. Hence, the 75 relevés were divided into UPA (36 plots, black circles) and PA (39 plots, green diamonds) for further ordination.

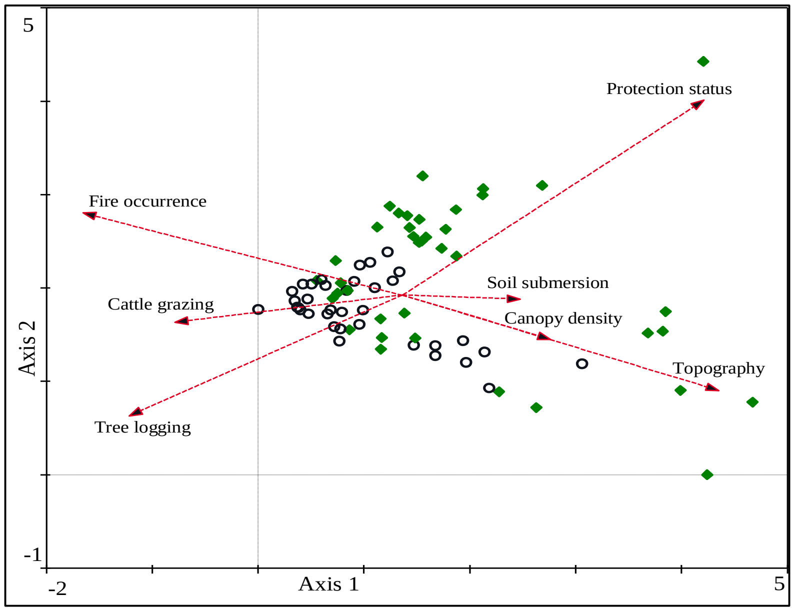

Figure 4.1. DCA of 75 relevés x 142 woody species in PA (diamonds) and UPA (circles)

a) Plant communities and biophysical conditions in UPA

The ordination of the 36 relevés x 100 species in UPA (Figure 4.2) showed four vegetation types (U1, U2, U3, U4, U stands for UPA). Appendices 3 and 4 showed that the first two axes of the DCA (Figure 4.3) defined gradients of soil moisture conditions and topographical positions**.** Axis 1 defines plant communities of moist and finer soils in lowlands (U1) versus those of dry and rocky soils located on relatively high lands (U2, U3 and U4). The DCA plot of environmental variables indicated a gradient of canopy cover density related to increasing soil moisture content and topographical position (axis 1) while axis 2 denoted mostly the effects of anthropogenic factors. The discriminated four main clusters were:

- Group U1, composed of eight (8) plots, included only riparian and dry forests located on riparian lands. Most of these plots occurred in gullies, riverbanks and inland valleys with relative gentle topographical positions. It was the cluster of plant communities growing on submersible soil with high levels of water and moisture content, due to their adjacency to water lines. The most dominant woody species were *Vitex doniana*, *Diospyros mespiliformis, Pterocarpus santalinoides,* and *Anogeissus leiocarpus.*
- Group U2 made up of seven plots, consisted of woodlands/open forest stands, of which some are highly threatened by human disturbances. This group occured on the low-slope and flat terrain. Some of the stands were naturally less dense due to topographical positions with rocky soils, which did not favour seed germination. The most frequent species were *Vitellaria paradoxa*, *Diospyros mespiliformis*, *Daniella oliveri* and *Pterocarpus erinaceus*.
- Group U3 (11 plots) comprised tree savannahs and some highly degraded woodlands. They occurred on mid-slopes and some flat terrains of medium altitude. Similar to U2 and U4, the common environmental threats occurring in these communities and those of U2 and U4 were the high level of wood extraction (charcoal production and tree logging), fire occurrence, and cattle grazing. *Daniella oliveri, Isoberlinia doka, Pterocarpus erinaceus, Vitellaria paradoxa*, *Parinari congensis* were the most frequent species in this group.
- Group U4 constituted mainly by shrubby savannahs, some scattered tree savannahs located mainly on top-slopes, and some plateaus with rocky soils. It comprised 10 plots with high fire imprints indicating the low level of tree canopy cover associated with the high proportion of grasses and bushes. Some typical species of these savannahs were *Pteleopsis suberosa*, and *Detarium microcarpum.* The most frequent species were

Detarium microcarpum, Vitelaria paradoxa, Pteleopsis suberosa, Sarcocephalus latifolius, and Parinari congensis.

From the combined analysis of the above-mentioned DCA plots in UPA, it was observed that U1 occured mostly on richer soils with high topsoil OM. High canopy cover, low slope and high soil moisture (TWI) characterised U1 though human pressures tended to modify the natural patterns. The horizontal axis denotes a gradient of human pressures increasing from the left to the right. Groups U1 and U4 were opposites, with U1 in lowland and U4 on top-slope. The relative high number of plots occurring in the right part of the DCA indicated that the majority of the vegetation types experienced human pressures such as illegal logging and grazing, and bush fires.

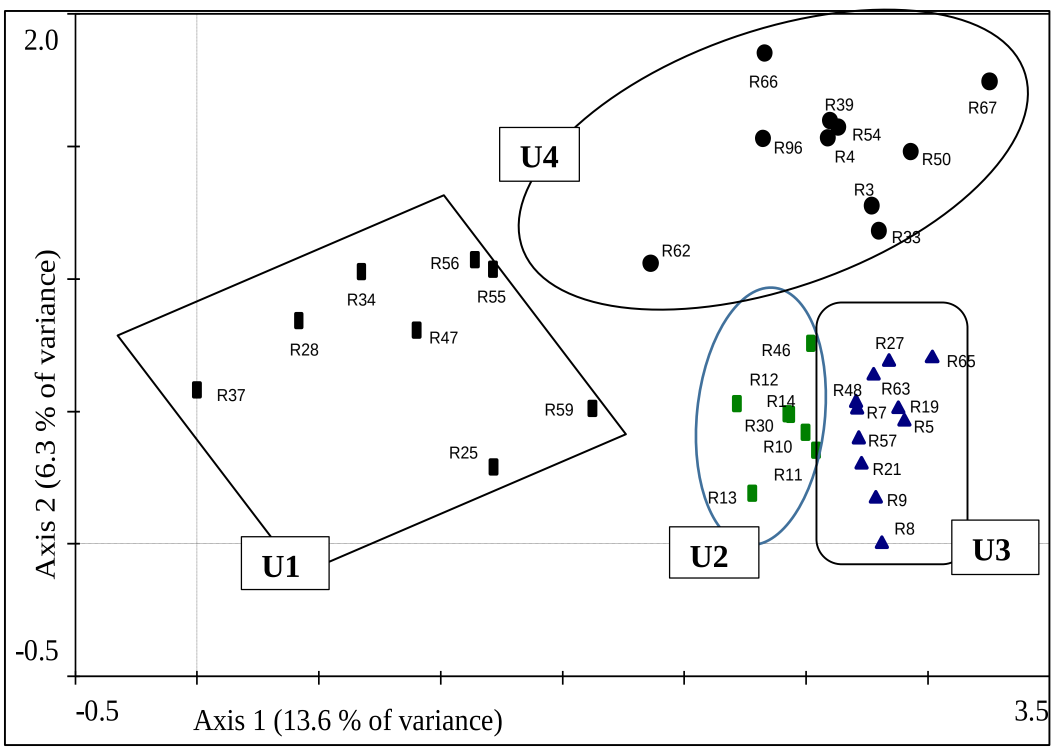

Figure 4.2. DCA of 36 plots and 100 woody species recorded in UPA

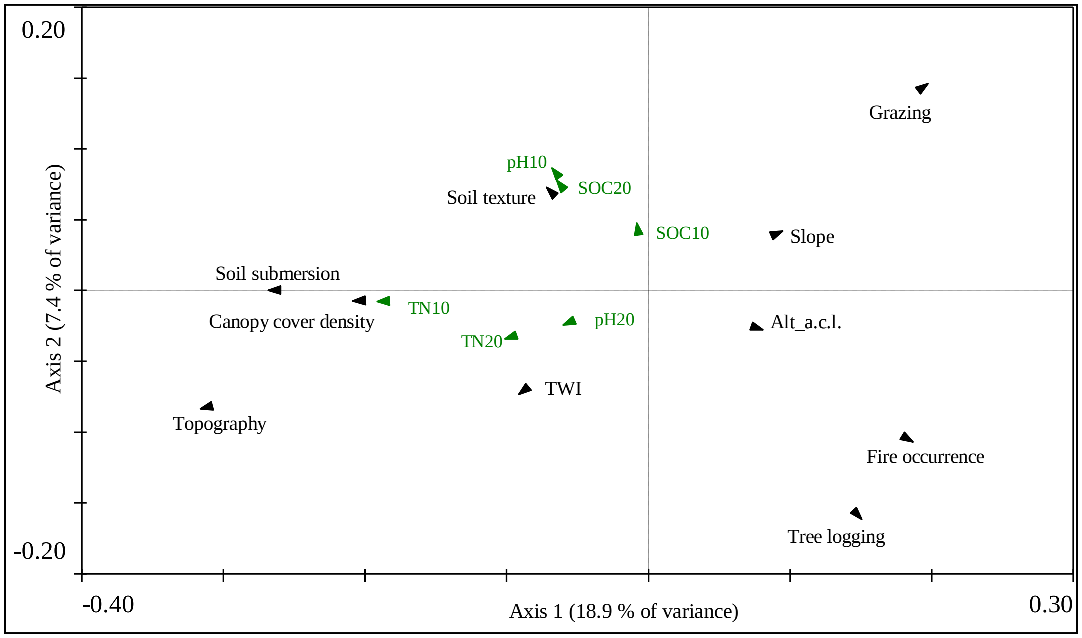

Figure 4.3. DCA showing biophysical and soil properties under the 36 plots in UPA

b) Plant communities and biophysical conditions in protected areas

The DCA for PA samples (Figure 4.4) showed three vegetation types clustered. Axis 1 exhibited a gradient of soil nutrient contents in relation with topography, which correlated positively with this axis (Appendices 5 and 6). The low cumulative variance indicated that most of the variance in the information was scattered along other components of the DCA, suggesting that there were intrinsic features to each site despite the similar conditions defining the vegetation groups. However, pH and SOC in subsoil showed a negative correlation with the axis 1. Thus, axis 1 indicated a gradient of soil nutrient richness defined by topography and affected by wildfire occurrence. The DCA plot of environmental and biophysical variables and human traits (Figure 4.5) described the three groups P1, P2, and P3 (P stands for PA) in the vegetation of PA.

- P1 (10 plots) was a mosaic of riparian vegetation comprising mostly gallery forests. Most of these plots occurred in gullies, valleys and relatively gentle topographical positions. It was the group of lowland forest stands growing in conditions of high water level and moisture availability. The most dominant woody species were *Vitex doniana*, *Diospyros mespiliformis*, *Pterocarpus santalinoides*, and *Anogeissus leiocarpus.*
- P2 was made up of 10 plots of woodlands/open forests. Some of these stands were highly threatened by human *disturbances*. This group comprised stands occurring on mid-slopes and flat terrains. Some of the stands are naturally density-low due to topographical positions (hillsides) with rocky soils, which do not favour seed germination. The most frequent species occurring in these stands were *Vitellaria paradoxa*, *Diospyros mespiliformis, Daniella oliveri* and *Pterocarpus erinaceus*.
- P3 (19 plots) comprised tree savannahs and some degraded woodlands. The common environmental threats were the high level of wood extraction (charcoal production and tree logging), fire occurrence, and cattle grazing. *Daniella oliveri*, *Isoberlinia doka*, *Pterocarpus erinaceus, Vitelaria paradoxa and Parinari congensis* were the most frequent species of this group. The most frequent species are *Detarium microcarpum, Vitelaria paradoxa, Pteleopsis suberosa, Sarcocephalus latifolius,* and *Parinari congensis.*

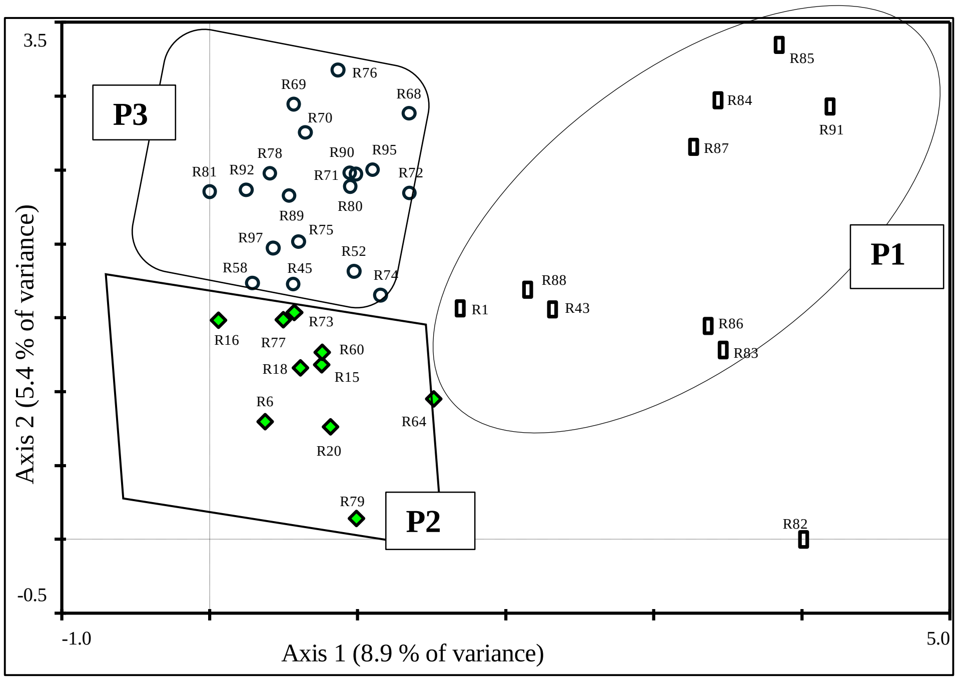

Figure 4.4. DCA of 39 plots and 121 woody species in PA

Combined analyses of the above DCA plots in PA showed the coexistence of the biophysical and human threats that shaped landscape composition. Indeed, P1 regrouped sites with topsoils richer in SOC and TN, and subsoils in TN. P1 was often at low topographic sites with high soil moisture (high TWI). Meanwhile, P2 occurred on soil with high pH and rich sublayers. They often experienced more tree logging because of their relative low slope and their canopy cover density. However, because of the relative low canopy cover density in P3, fire and grazing did occur often in these plots because of their richness in grasses and herbaceous fodder. In sum, the horizontal axis (axis 1) was associated with an increasing gradient of topsoil nutrients and soil moisture from the right to the left side (summits and mid-slopes versus lowlands and river banks). A combined effect of increasing gradient of human pressures from right to left was also observed from this plot (highly disturbed vegetation versus less disturbed vegetation). However, the second axis indicated mostly a decreasing gradient of vegetation physiognomy, which was closer from negative to positive canonical axes.

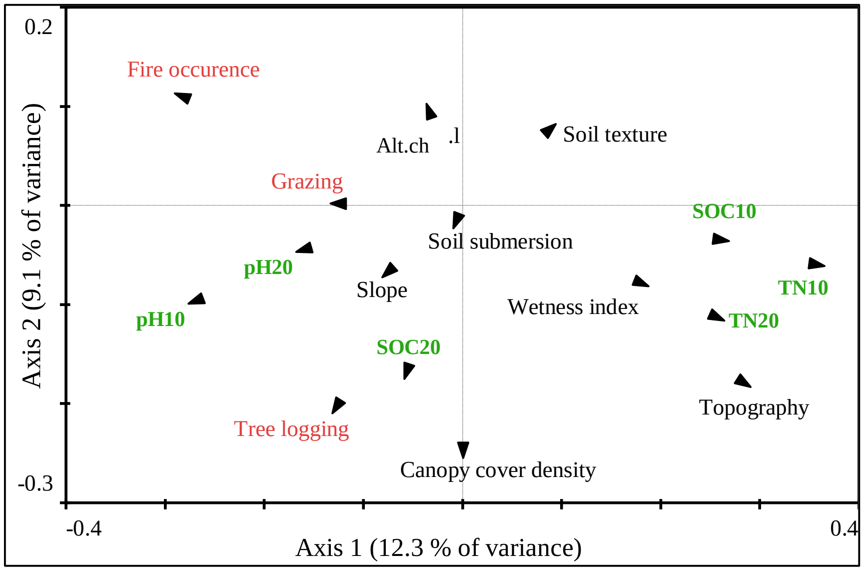

Figure 4.5. DCA showing biophysical and soil properties in the sample sites of PA

4.3.3. Structural and dendrometric characteristics of plant communities of Mo basin

Tables 4.1, 4.2 and 4.3 show the **s**tand characteristics and diversity indices of the various plant communities in free-access lands, protected areas**,** and merged analyses for PA versus UPA, respectively**.** Appendix 7 provides an overview of the 3-parameters Weibull distribution for the height and diameter distribution in the different plant groups.

In UPA, stands characteristics exhibited significant differences among groups (*ANOVA*, $p < 0.05$), except for mean species richness (*ANOVA*, $p = 0.362$), tree density with saplings (*ANOVA*, $p = 0.580$), and sapling density (*ANOVA*, $p = 0.509$). Forested stands clustered in group U1 showed the highest mean values of tree height (10.5 ± 6.3 m), diameter (28.0 ± 17.4 cm), and density (947 ± 328 stems ha^{-1}). With a mean basal area of 58.8 ± 14.1 m^2 ha^{-1}, U1 exhibited the highest record, compared to U2, U3, and U4 with 18.6 ± 6.2 m^2 ha^{-1}, 15.9 ± 7.7 m^2 ha^{-1}, and 7.7 ± 3.3 m^2 ha^{-1}, respectively.

Table 4.1. Stand characteristics and soil properties in the vegetation types of UPA

Characteristics	Cluster U1 (8 relevés)	Cluster U2 (7 relevés)	Cluster U3 (11 relevés)	Cluster U4 (10 relevés)	ANOVA ($\alpha = 0.05$)
Species richness	58	51	66	49	*na*
Mean species richness	18.75 ± 4.49 A	20.14 ± 5.61 A	19.27 ± 5.35 A	16.20 ± 4.02 A	*0.362 ns*
Mean height (m)	10.50 ± 6.32 A	6.72 ± 4.15 b B	5.43 ± 3.68 C	5.04 ± 3.46 C	*0.000 ss*
Mean DBH (cm)	28.02 ± 17.42 A	19.28 ± 8.68 B	18.32 ± 9.28 B	19.35 ± 11.15 B	*0.000 ss*
Mean diameter with saplings (cm)	21.79 ± 17.61 A	13.22 ± 9.08 B	10.57 ± 8.14 C	8.82 ± 7.96 D	*0.000 ss*
Basal area G (m^2/ha)	58.80 ± 14.07 A	18.58 ± 6.19 B	15.93 ± 7.69 B	7.66 ± 3.33 C	*0.000 ss*
Tree density (trees/ha)	946.9 ± 328.2 A	920.6 ± 490.4 A	845.7 ± 431.8 A	692.2 ± 427.5 A	*0.580 ns*
Density without saplings (trees/ha)	678.1 ± 216.9 A	487.3 ± 166.7 B	311.4 ± 141.7 C	155.6 ± 100.5 D	*0.000 ss*
Sapling density (saplings/ha)	268.8 ± 218.9 A	433.3 ± 444.6 A	534.3 ± 483.9 A	536.7 ± 444.3 A	*0.509 ns*
Shannon index H' (bits)	5.038 ± 0.076	4.559 ± 0.095	5.071 ± 0.077	4.653 ± 0.090	*na*
Pielou evenness (unitless)	0.879	0.804	0.836	0.820	*na*

Note: One-way Anova outputs: na= Not available; ns= not statistically significant; ss= statistically significant. Values that do not share a letter are significantly different at 95% CI using anova with post-hoc test (p=0.05). Capitalized letters are the outputs from Tukey comparison test.

U3 was the species-richer group (66 species), exhibiting high diversity features (5.07 and 0.84, for *H'* and *E*, respectively) close to the values obtained for U1. The high *H'* and *E* are indicators of stability and homogeneity of studied landscapes. Saplings with the highest density (537 ± 444 saplings ha^{-1}) dominated U4 (Table 4.2), the record of low values. In this group U4, species richness was about 49 with mean diameter and height of 19.4 ± 11.2 cm and 5.0 ± 3.5 m, respectively. Saplings had a strong effect on mean diameters in all groups resulting in mean basal areas that decreased from U1 to U4 (Appendix 7).

Table 4.2. Stand characteristics and soil properties in the vegetation types of PA

Characteristics	Cluster P1 (10 relevés)	Cluster P2 (10 relevés)	Cluster P3 (19 relevés)	ANOVA $\alpha = 0.05$
Species richness	68	70	70	*na*
Mean species richness (per plot)	13.50 ± 3.78 A	16.20 ± 2.94 A	14.58 ± 3.73 A	*0.245 ns*
Mean height (m)	15.53 ± 6.53 A	10.62 ± 5.37 B	9.469 ± 4.087 C	*0.000 ss*
Mean DBH (cm)	28.01 ± 16.58 A	23.95 ± 14.71 B	20.51 ± 10.07 C	*0.000 ss*
Mean diameter with saplings (cm)	26.30 ± 16.88 A	20.83 ± 14.90 B	18.65 ± 10.55 C	*0.000 ss*
Basal area G (m^2 ha^{-1})	46.26 ± 31.77 A	32.55 ± 18.97 A	15.09 ± 7.99 B	*0.001 ss*
Tree density (trees ha^{-1})	604.0 ± 300.8 AB	712.4 ± 532.9 A	407.0 ± 201.3 B	*0.065 ns*
Density without saplings (trees ha^{-1})	560.0 ± 279.8 A	553.7 ± 296.9 A	356.1 ± 155.5 B	*0.036 ss*
Sapling density (saplings ha^{-1})	50.0 ± 72.0 A	163.2 ± 281.9 A	50.9 ± 97.0 A	*0.176 ns*
Shannon index H' (bits)	4.378 ± 0.096	4.605 ± 0.088	5.016 ± 0.078	*na*
Pielou evenness. Eq (unitless)	0.776	0.793	0.859	*na*

Note: One-way Anova outputs: na= Not available; ns= not statistically significant; ss= statistically significant. Values that do not share a letter are significantly different at 95% CI using anova with post-hoc test (p=0.05). Capitalized letters are the outputs from Tukey comparison test.

Table 4.3. Broad stand characteristics and soil properties in PA versus UPA

Characteristics	PA (39 relevés)	UPA (36 relevés)	ANOVA *(α = 0.05)*
Species richness	121	100	na
Mean species richness (per plot)	14.62 ± 3.54 A	18.47 ± 4.90 B	*0.000* ss
Mean height (m)	11.16 ± 5.63 A	6.41 ± 4.62 B	*0.000* ss
Mean DBH (cm)	23.21 ± 13.50 A	21.25 ± 12.62 B	*0.001* ss
Mean diameter with saplings (cm)	20.99 ± 13.79 A	12.46 ± 11.14 B	*0.000* ss
Basal area G (m^2 ha^{-1})	27.56 ± 23.04 A	23.67 ± 21.10 A	*0.450* ns
Tree density (trees ha^{-1})	535.8 ± 354.4 B	840.1 ± 416.2 A	*0.001* ss
Density without saplings (trees ha^{-1})	459.1 ± 247.4 A	383.8 ± 247.0 A	*0.192* ns
Sapling density (saplings ha^{-1})	79.4 ± 164.3 A	456.3 ± 416.0 B	*0.000* ss

Note: One-way Anova outputs: na= not available; ns= not statistically significant; ss= statitistically significant. Values that do not share a letter are significantly different at 95% CI using anova with post-hoc test (p=0.05). Capitalized letters are the outputs from Tukey comparison test.

On the other hand, plant communities in PA were characterized by relatively high values of the stand and diversity features compared to those in UPA. An exception was observed for sapling density and diversity indices *H'* and *E*. Apart from the average species richness per plot and the sapling density, all other features varied significantly according to plant communities. Mean DBH was about 28.0 ± 16.6 cm, 24.0 ± 14.7 cm, and 20.5 ± 10.1 cm in P1, P2 and P3, respectively. This trend in tree diameters resulted in similar trend in the basal areas with highest value in P1 (46.3 ± 31.8 m^2 ha^{-1}) and lowest in P3 (15.1 ± 8.0 m^2 ha^{-1}). Though the basal area and the tree density were the lowest in P3, this plant community exhibited high diversity of species (5.02 and 0.86 for *H'* and *E*, respectively). With a density of 712 ± 533 trees ha^{-1}, P2 was the densest vegetation type compared to P1 (604 ± 301 trees ha^{-1}) and P3 (407 ± 201 trees ha^{-1}). Basal areas in the PA were more shaped by the large contribution of mature individuals than saplings which exhibited low densities, especially in P1 (50 saplings ha^{-1}) and P3 (51 saplings ha^{-1}).

A broad analysis showed that most stand features and diversity differed significantly between PA and UPA. PA showed higher values of stand features, except tree density and sapling density. Only basal area and tree density without saplings were not significantly different between the two groups. There were 122 and 100 woody species recorded in PA and UPA, respectively. Mean sapling density was very low in PA (79 saplings ha^{-1}) compared to those in UPA (456 saplings ha^{-1}). This was probably due to low potential of vegetative multiplication through natural process (suckering and seedlings) in PA. On the average, trees in PA were taller (11.2 ± 5.6 m) and bigger (23.2 ± 13.5 cm) than those in UPA (6.4 ± 4.6 m and 21.3 ± 12.6 cm, respectively for height and diameter).

Based on stand characteristics, no significant similarity existed between the seven groups, except between U2 and U3 ($I_J = 0.539$) (Table 4.4 and Figure 4.6). Closest groups in similarity were P2 - P3 ($I_J = 0.458$), U1 - U2 ($I_J = 0.473$), U2 - U4 ($I_J = 0.493$), and U3 - U4 ($I_J = 0.456$). The recorded similarity values indicated substantial common characteristics among groups ($I_J \geq 0.333$ for most of them).

Table 4.4. Similarity between discriminated vegetation stands

Vegetation groups	P1	P2	P3	U1	U2	U3
P2	0.366					
P3	0.289	0.458				
U1	0.326	0.438	0.333			
U2	0.202	0.441	0.407	0.473		
U3	0.186	0.447	0.432	0.333	**0.539**	
U4	0.206	0.368	0.384	0.372	0.493	0.456

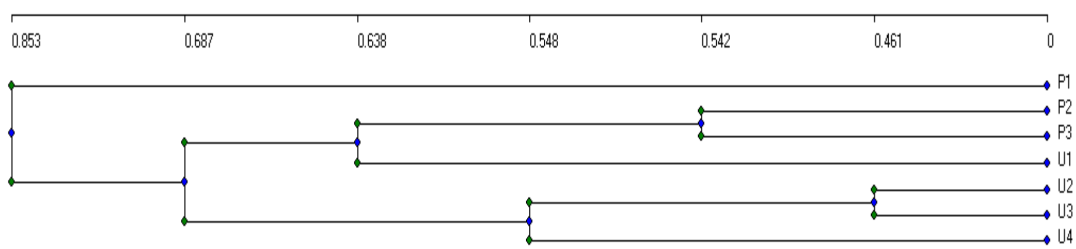

Figure 4.6. Similarity between the vegetation groups based on stand characteristics

4.3.4. Ecology and human disturbances within vegetation types

The analyses of ecological features and human impacts in the different vegetation types (Table 4.5) indicated differences related to *in-situ* conditions. In general, soils of the seven vegetation types are acidic (pH < 7) with relative high SOC and TN in the topsoil (0 - 10 cm). Though pH and SOC for the 20 cm depth did not vary significantly among groups ($p > 0.05$), other ecological variables did. Based on ecology and human footprints, dendrogramme (Figure 4.7) indicated four clusters: (1) U1 and P1 were located on soils with high SOC and TN at riverbanks and inland valleys. Soil waterlogging is higher than 30 %. For both U1 and P1, grazing and tree logging were of low rates. On average, mean Alt.ch of 7.2 m is observed in U1 versus 11.2 m for P1, resulting in high density of canopy in U1 (coefficient of 3.00 versus 2.10). (2) Vegetation of U2 carried some particular stands which exhibited relatively nutrient-rich soils with less moisture content (33.2 m above channel). (3) U3 and P2 developed on soils with medium nutrient content associated with high rate of illegal tree logging (0.70 in P2) and fire occurrence (0.70 and 0.91 in P2 and U3, respectively). This group was the timber-rich stands, explaining the high rate of tree logging, even in PA. Their soils had low waterlogging related to their high Alt.ch (24.4 m for U3 and 25.1 m for P2). (4) U4 and P3 are relevés with less nutrient contents experiencing moderate human disturbances, especially in U4. On average, they occur at the same Alt.ch (19 m). The soils were less waterlogged with low canopy density favourable for grazing (coefficients of 0.70 and 0.79) but not for tree logging. The number of plant communities in PA was due to the less disturbed vegetation which facilitated easy demarcation of the three vegetation types. However, in the UPA, human effects changed the physiognomy of some stands and increased the patterns defining the vegetation types.

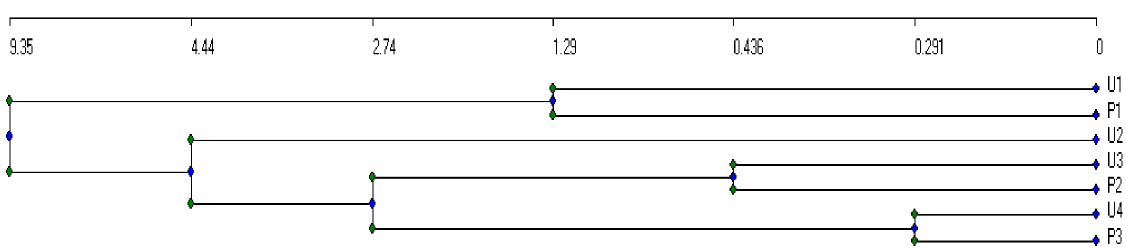

Figure 4.7 Similarity between vegetation groups based on ecology and human factors

Table 4.5. Soil chemical properties and ecological conditions in the plant communities in both UPA and PA

Groups	Plots	pH		TN (in %)		SOC (in %)		SWL	Bushfire	Canopy	TreeL	Grazing	Alt.ch
		0-10 cm	10-30 cm	0-10 cm	10-30 cm	0-10 cm	10-30 cm	Site level	Site level	Site level	Site level	Site level	Site level
U1	8	6.41±0.39 A	6.19±0.58 A	0.133±0.08 B	0.071±0.04 BC	2.44±1.29 BC	2.31±0.77 A	0.75 A	0.50 BC	3.00	0.13	0.00	7.17
U2	7	6.31±0.26 AB	6.03±0.13 AB	0.119±0.02 BC	0.074±0.01 BC	3.09±0.96 AB	2.05±0.53 AB	0.14 B	1.00 A	1.86	0.57	0.14	33.15
U3	11	6.23±0.23 AB	6.16±0.57 A	0.081±0.03 C	0.057±0.02 BC	2.21±0.69 C	1.93±0.57 AB	0.09 B	0.91 A	1.45	0.64	0.36	24.36
U4	10	6.48±0.49 A	6.03±0.45 AB	0.077±0.04 C	0.052±0.02 C	2.40±0.71 BC	2.11±0.99 AB	0.10 B	0.70 AB	1.30	0.30	0.60	19.52
P1	10	5.95±0.45 B	5.70±0.28 B	0.215±0.09 A	0.101±0.04 A	3.63±1.46 A	1.72±0.43 B	0.30 B	0.20 C	2.10	0.10	0.00	11.24
P2	10	6.35±0.17 A	6.29±0.50 A	0.129±0.05 B	0.077±0.04 AB	3.01±0.68 AB	1.84±0.46 AB	0.20 B	0.70 AB	2.40	0.70	0.00	25.07
P3	19	6.36±0.37 A	6.01±0.34 AB	0.104±0.04 BC	0.062±0.02 BC	2.55±0.56 BC	1.72±0.39 B	0.37 B	0.79 AB	2.00	0.11	0.16	19.36
Anova at p = 0.05		0.047 ss	0.086 ns	0.000 ss	0.002 ss	0.010 ss	0.242 ns	0.024 ss	0.002 ss	0.004 ss	0.001 ss	0.001 ss	0.543 ns
Overall U	36	6.35±0.37	6.10±0.47	0.099±0.05	0.062±0.02	2.48±0.92	2.09±0.73	0.25	0.78	1.83	0.42	0.31	20.90
Overall P	39	6.25±0.39	6.00±0.42	0.139±0.07	0.076±0.03	2.94±0.98	1.75±0.42	0.31	0.62	2.13	0.26	0.08	18.74
Anova at p = 0.05		0.247 ns	0.325 ns	0.009 ss	0.038 ss	0.041 ss	0.015 ss	0.584 ns	0.131 ns	0.207 ns	0.145 ns	0.011 ss	0.733 ns

Note: One-way Anova outputs: na = not available; ns = not statistically significant; ss = statitistically significant. Values that do not share a letter are significantly different at 95% CI using anova with post-hoc test (p = 0.05). Capitalized letters are the outputs from Tukey comparison test. SWL = Soil waterlogging; TreeL= Tree logging; Grzing = animal grazing

4.4. Discussion

4.4.1. Vegetation patterns, structure and dynamics

At the landscape level, the Mo basin composed of a wide range of natural and human-influenced ecosystems. Apart from productive managed landscapes (farms, fallows and pastures), dry forests, gallery forests, woodlands, tree/shrub savannahs are the common cover types defined according to soil conditions in relation to topography as well as species composition. These biophysical variables determine the physiognomy of the vegetation types which stand characteristics and soil conditions differed significantly (Dourma, 2008; Wala et al., 2012). Along the topographical gradient denoting soil moisture conditions, canopy cover was denser in lowlands (inland valleys, low-slopes and river banks) than mid-slopes and top-slopes, indicating that the wetness index plays an important role in species composition and vegetation growth (Aynekulu, 2011). Though the stand characteristics did not really help in defining the vegetation types, it was evident that larger trees barely occurred in top-slopes dominated by tree savannah and shrubs, because of the coarser and rocky soil conditions, which do not favour soil moisture for plant growth. This consequently resulted in the stand basal area values, which showed a decreasing trend from lowlands to top-slopes dominated by shrubs/ tree savannahs. Along the protection status gradient, mean basal area (27.6 m^2 ha^{-1}) was higher than that of the UPA (23.7 $m^2 ha^{-1}$) but included in the range of values recorded by studies in other similar landscapes made up of subtropical dry forests and woodlands (23.8 - 78.8 m^2 ha^{-1}) (Dourma et al., 2009; Folega et al., 2012; Wala et al., 2012). These results indicated a more pronounced exploitation of the tree species in UPA.

In general, the poor representation of saplings could be due to the rocky nature of the soils that did not favour quick germination of seeds, and to landscape roughness inducing seeds transportation downslope and their being washed away by water. This could explain the high density of saplings in unprotected areas, which lie, mostly on less rough landscapes. On the other hand, the prevalence of tall and big trees in protected lands compared to UPA suggested the effects of intensive and selective tree logging, charcoal production and mortar making on vegetation structure. These impacts often target specifically high and big trees, inducing the loss of trees of big size. Subsequently, human pressures convert tree-rich into tree-less stands and induce savanisation. The PA showed a more stable landscape with a bell-shaped distribution of tree densities (Appendix 7). This could be because trees of big sizes were subjected to selective logging in UPA, increasing therefore the small stem density in UPA compared to PA.

4.4.2. Relationships between vegetation types and environment

Ecology and biophysical factors usually define vegetation types, which, in-turn, contribute to the maintenance of these factors, especially edaphic-ecological variables. As shown by previous studies, the distribution of plant community or vegetation types is defined along environmental gradients such as topography, soil conditions, microclimate, and human disturbances. In this study, whatever the protection status, vegetation types were mostly defined according to topographical gradient inducing different soil conditions (moisture and nutrient contents). Riparian and dry forests and woodlands occurred on moist soils with finer particles and high nutrient contents. These findings accord with those of former authors (Dourma, 2008; Wala et al., 2012; Woegan, 2007) who worked in the same zones. Regarding soil nutrients and particles, the flow gradient from top-slopes to low-slopes and inland valleys induced a typical gradient of soil conditions. Naturally, each land cover occurs on particular soil physico-chemical properties which highly vary spatially, even within the same LUC type (Wiesmeier et al., 2014b). Accordingly, landscape positions and elevation induced the spatial variability of soil conditions (Jabeen & Ahmad, 2009; Solon, 2007; Zhang & Zhang, 2010) controlled by the hydrological processes and on the vegetation patterns through their effects on soil conditions (Aynekulu, 2011). Soils in lowlands and low-slopes did have deeper profiles with high moisture and nutrient content favourable for the plant growth. In contrast, top-slopes and steeper mid-slopes have drier, rockier, and less deep soil conditions. These ecological indicators explain the occurrence of forests along riversides and in lowlands.

Besides, anthropogenic disturbances affect species diversity, size classes and LUC types through habitat loss and fragmentation. Several studies indicate that human disturbances usually have substantial effects on ecosystem functioning, and therefore their structure, physiognomy and species composition (Appiah, 2013; Dourma et al., 2009; Folega et al., 2010; Ouedraogo, 2010; Paré, 2006). Human footprints caused by grazing, tree logging, wildfire, and cropping have been reported as negative drivers inducing habitat fragmentation with less tree species diversity and loss of landscape functions and aesthetics, even in PA (Appiah, 2013; Tchabsala & Mbolo, 2013). The induced LD also has subsequent effects on soil conditions, especially nutrient loss, carbon and nitrogen cycles (Traoré et al., 2015).

In addition to land protection status, topography and soil conditions were the most prominent variables defining the landscape patterns i.e. species composition and vegetation structure and dynamics. It often appears that landscape fragmentation and deforestation largely occur in accessible areas such as lowlands. In these accessible areas, the transformation of wild landscapes (forests and woodlands) into other land-use types (farms and pastures, plantations) is a result of people livelihood support activities (Appiah et al., 2009; Pare et al., 2010). It is suggested that land conservation measures should target these vulnerable areas in order to ensure effectiveness and efficiency of implementation. As the PA showed some weaknesses in land conservation due to inefficient management regime and law enforcement, it is important to investigate alternatives for better management (Appiah, 2013; Damnyag et al., 2013; Wala et al., 2012). Besides, wild UPA, especially in areas with low population density, could be of great importance in biological conservation, even if an explicit role of conservation is not devoted to them.

4.4.3. Implications for sustainable land management and landscape restoration

With regard to the above-mentioned issues, land management in the Mo river basin should evolve adapted strategies that define clearly property rights, reinforce laws and policies, and involve all stakeholders. These strategies, involving all stakeholders are based on current rethinking of collective management of common resource pools, especially forests and lands, to avoid "tragedy of commons". Governance and management systems combined with socioeconomic conditions of people being the underlying factors in decision-making regarding land use (Kaye-Zwiebel & King, 2014; Specht et al., 2015), law enforcement for PA without any implication of local stakeholders will guarantee failure in sustainable conservation strategies. In addition, agroforestry appears to be of great potential for sustainable development and climate mitigation (Mbow et al., 2014a), especially using native tree species of local economic importance such as *Vitellaria paradoxa* and *Isoberlinia spp* (Dourma et al., 2009). The quasi-stability of human-affected landscapes and ecosystems is an indicator that UPA can gain more attention to ensure adapted land use/conservation (Ellis, 2013; Gu & Subramanian, 2014).

4.5. Conclusion

This study helped in the identification of the linkage between social-ecological systems, and the landscape configuration under different land protection status. *in-situ* ecological conditions, soil information, land protection status and human disturbances define 4 vegetation cover types (forests, healthy and degraded woodlands, and savannahs/shrubs) in UPA with high human disturbance indices; and 3 cover types (forests, woodlands, and savannahs/shrubs) in PA with indicators of human encroachment. In general, the similarity level among the seven vegetation types is very low (< 50 %). Their soils were acidic (pH < 7) with high contents of SOC and TN in the topsoil (0 - 10 cm). Land protection status was an important factor affecting and shaping the vegetation physiognomy in the Mo basin. Accordingly, the common environmental threats were the high level of wood extraction (firewood, charcoal production and tree logging), bush fires, and cattle grazing. Though the natural biophysical factors shape landscape physiognomy, human disturbances affect the structure and composition of plant communities through the provision of multiple and valuable ESS to local people. The study showed that biological conservation should not only target landscapes in PA but also on some wild landscapes located in inaccessible and low populated areas. Efforts to restore degraded ecosystems and to promote sustainable landscapes have been also demonstrated throughout this comprehensive analysis of soil-vegetation relationships in a spatially explicit way at landscape level.

CHAPTER 5: ANALYSES OF THE HISTORICAL LAND USE/COVER CHANGES IN THE MO RIVER BASIN [3]

5.1. Introduction

Monitoring land resource dynamics for global change mitigation continue to be of grave importance at global scale as well as the national and local levels. Estimates in the FAO report indicated that forest areas have decreased by about 3.1 % over the past 25 years with the greatest deforestation and forest degradation occurring in tropical areas of America and Africa (FAO, 2015). The same report indicated that between 2010 and 2015, the annual gain of forestlands was far below the extent of loss, resulting in a net annual loss of 3.3 million ha of forests per annum. Regardless of scale, LUCC occur continuously, making it difficult to monitor their extent and quality. Yet, regular assessment and monitoring landscape dynamics is an essential step for the real understanding of change drivers and for changing mindsets towards sustainability (Houet et al., 2010).

Spatio-temporal changes of landscapes are continuous processes fully maintained by both natural and human-related drivers. Human imprints on terrestrial ecosystems are of the significant extent and of major environmental concern (Ellis, 2011; Gaia, 2011) relatively to habitat fragmentation, lands and ESS decline, biodiversity loss, livelihood decrease and climate variability (Balthazar et al., 2015; Ellis, 2013; Schleuning et al., 2011). The main drivers of these changes include large-scale forest exploitation, agricultural deforestation and small-scale forest disturbances (Damnyag et al., 2013; Lambin et al., 2003; Specht et al., 2015). These increasing pressures affect the resilience,

[3] This chapter is a revised manuscript accepted for publication in *Journal of Geographical Sciences (Springer)*

conservation efforts and the capacities of land resources to provide ESS, even in PA (Damnyag et al., 2013; Vedeld et al., 2012).

The use of satellite archives such as Landsat data have brought new insights into the approaches of understanding of land use/cover change (LUCC) and monitoring deforestation and forest degradation (DFD). The successes of the application of RS and GIS in the assessment of landscape dynamics reside in the availability of earth observation data and the multitude of methods for LUCC mapping and assessment (He et al., 2014; Kim et al., 2014; Zheng et al., 2014; Zhou et al., 2014a). These data and methods offer great potential to cover various spatio-temporal scales of analyses and monitoring of LUCC and the processes of landscape dynamics (Farooq, 2012; Rogan & Chen, 2004). Thus, the integration of multi-temporal satellite data with GIS and field data showed great insights in analysing landscape dynamics at various scales.

In Togo, several recent studies have been undertaken using RS and GIS to assess and monitor the changes in land resources at national and local scales (Adjonou et al., 2010; Badjana et al., 2014; Folega et al., 2015; Folega et al., 2014b). Human activities especially agricultural expansion, illegal tree logging and incursions in protected areas as well as settlement enlargement in rural and semi-urban areas have been identified as responsible for most of the changes (Dourma et al., 2009; Fontodji et al., 2011; Kokou et al., 2009). These studies have shown that current trends of land resources do not favour the functional services of the different ecosystems. Therefore, continual and complementary studies are encouraged to deepen the knowledge on the processes and determinants of LUCC in order to support integrated landscapes. Such studies have not received much research attention in the Mo river basin with its numerous social and ecological interests (Dourma et al., 2009). Yet, timely monitoring of LUCC processes is needed to provide the requisite data for decision-makers to have up to date information on LD issues. Using Landsat archives the objective of the study in this chapter is to assess the LUC dynamics in the Mo river basin over the period 1972 – 2014 in order to develop land monitoring information that facilitates strategies for adapted land management and rural development.

5.2. Materials and Methods

5.2.1. Study area

a) Population and income-generated activities

The population is mainly composed of rural farmers and cattle herders. According to the latest census data, the Mo river basin had about 17,761 inhbts with a density ranging from 0 to 250 inhbts km^{-2} (Figure 5.1) dominated by the *Tem* ethnic group (DGSCN, 2010). The same census indicated that the central region embedding the Mo basin had the lowest population density (47 person km^{-2} compared to 109 person km^{-2} for national average).

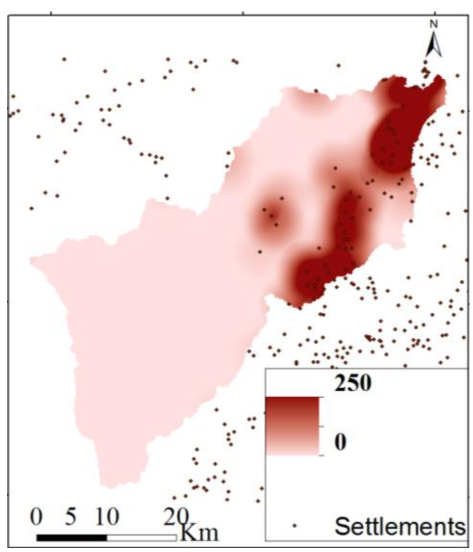

Figure 5.1. Population density (persons km^{-2}) in the Mo basin

Source: author mapping based on census data (DGSCN, 2010)

As far as income generated activities are concerned, agriculture is the most important activities occupying almost all the population. Agriculture is rainfed and mainly for subsistence. Main crops are yam, rice cassava, maize, millet, sorghum, groundnuts and beans. Some orchards of mango trees, cashew plantations, and citrus fruits are income sources. In addition, animal husbandry (poultry, capra, cattle, sheep, etc.), fishing, honey harvesting, hunting are common practices to support farming-based resources. People in the Mo basin are also involved in commercial charcoal and wood production.

b) Biomass energy and development

At the national level, the estimated fuelwood consumption was about 1756.09 Mg in 2000 while it increased to 2130.36 Mg in 2008 (DGE, 2010). This biomass energy is derived from natural landscapes as fuelwood and charcoal from many regions including the central region (Dourma et al., 2009; Kokou et al., 2009; Fontodji et al., 2011). This energy consumption poses potential threats to land conservation efforts in the local areas, especially the protected areas (Wala et al., 2012; Dourma et al., 2009). The current situation of land use with regard to human incursions in some PA, especially FMNP, led to the redefinition of PA boundaries by the national government (MERF-Togo, 2013).

5.2.2. LUCC mapping for the period 1972 – 2014

a) Data sources and processing

Land-use and cover maps for 1972, 1987, 2000 and 2014 were obtained from the classification of historical Landsat images covering the Mo river basin. Single ortho-rectified images of Landsat 8 (03 December 2014), Landsat ETM+ (04 December 2000) and Landsat TM (30 October 1987) free of cloud were collected at the path 193/row 054 (https://earthexplorer.usgs.gov). A pair of Landsat MSS (10 November 1972) were downloaded from the same source at the path 207/row 053 and path 207/row 054. These data were acquired for the time corresponding to the onset of the dry season (October to December), enabling the clear distinction between LUC types, especially agricultural fields and typical savannahs in the landscape (Ruelland et al., 2010; Traore et al., 2014).

Developing land use and cover maps involved the process of clustering and assigning similar pixels into classes (Rojas et al., 2013). To reduce the effects of typical similarities between closer cover types such savannahs and woodlands, which look similar in the savannah-dominated landscapes, a set of transformation, was necessary. First, since the study was interested in vegetation cover mapping, NDVI was computed as independent layer not only to reduce the effects of topography but also to measure the distribution of vegetation health over the landscape of interest (Braimoh & Vlek, 2004b). NDVI is widely used as a powerful indicator of vegetation greenness, and less sensitive to topographic factors in mountainous areas (Diallo et al., 2010; Matsushita et al., 2007). Original bands were combined with the NDVI layer to perform the pixel-based supervised classification using Maximum Likelihood algorithm in ENVI 4.7 image processing software. Though topography is a common source of biases in LUC classification in mountainous areas, elevation were not integrated in the classification process, as the maximal elevation above sea level which is around 850 m, does not really provide significant hill shade at the sensor passing time (Diallo et al., 2010). The combined layers were registered on UTM WGS 84 projection system and used to extract spectral signatures for the classification of the respective images (Braimoh, 2004; Gutiérrez Angonese & Grau, 2014; Wittig et al., 2007).

Six main categories (Figure 2) were defined based on the national vegetation map (Afidégnon et al., 2003), complemented by the Land Classification System of FAO in order to better consider the physiognomy-structural conditions (vertical and horizontal arrangements as well as land use affected to the cover types) of the following types:

(1) Forests: close canopy vegetation including the riparian forests along streams and dry forests in lowlands. The canopy cover exceeds 60 % with an understory layer.

(2) Woodlands: open canopy vegetation including woody savannahs and woodlands. The canopy cover comprises between 30 to 60 % and do not possess understory vegetation making their cover less thick compared to forests. Trees are higher than 5 m.

(3) Savannahs: Treeless open canopy vegetation composed of tree savannahs, shrubs, and scattered grasslands. Generally, tree height is lower than 5 m. Including old fallows (> 3 years), they are bush or grass dominant with woody cover less than 30 %;

(4) Agricultural land: cultivated (including cereal crops, vegetable crops and fruit orchards) and non-cultivated (farm fallows less than 3 years and parklands) lands;

(5) Built areas: areas occupied by residential settlements as well as paved surfaces.

(6) Water: surface water bodies including rivers and reservoirs.

Paved surfaces and rocks are mostly confused among settlements and agricultural lands.

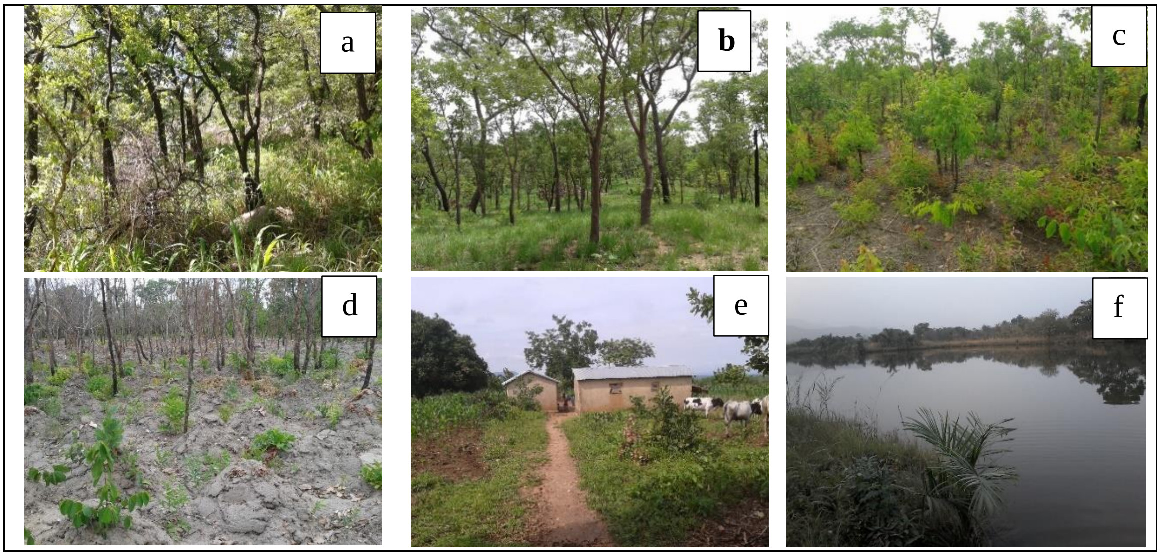

Figure 2. Different LUC units in the Mo basin: (a) Forest; (b) Woodland; (c) Shrub/Savannahs; (d) New farm; (e) Farmhouse; (f) Water body.

For each of the above mentioned land/use cover class, training areas were developed independently and the spectral characteristics of each training sample were checked through the separability tests of Jeffries-Matusita and Transformed Divergence (Braimoh, 2004; Zhou et al., 2008). The outputs of the separability tests range from 0 to 2, with 0 indicating poor separability and 2 a total separability i.e. the signatures have no similarity. For this study, the separability between the LUC types was good (Appendix 8).

Collection of reference data to assess the accuracy of historical maps is often a challenging issue in data scarce areas (Wilson & Sader, 2002). While the collection of such data is less hectic for recent and current images, it is very difficult for images of long history especially due to the lack of reliable data such as aerial photographs (Biro et al., 2013; White et al., 2013; Zhou et al., 2008). Therefore, the data dearth for the current study constrained the use of different sources of information. Ground truth information for accuracy assessment of classified map of 2014 relied on 177 field-registered GPS coordinates were randomly collected. These points were collected during field studies between November and May 2014 corresponding to the dry season and matching the acquisition season of the image. Validation samples of at least 45 pixels were composed of either the raw GPS points or a blend with homogenous polygons around the GPS points. For the validation of LUC map of 2000, the available topographic at 1/200,000 (IGN, 1986) and vegetation map of Togo (Afidégnon et al., 2003) were the reference documents combined with the LUC maps of 2005 and 2009 from GlobCover project (Bicheron et al., 2008). The reclassification of Globcover images was done to meet the classification system used in the current study. Historical Google Earth images were helpful in the creation of this validation information. For the early date images (1987 and 1972), homogeneous areas were selected to create representative validation samples based on the detection of unchanged areas (persistent pixels) along the time series. Using the geographic link tool from ENVI software, the validated maps of 2000 and 2014 were overlaid with each of the classified maps for 1987 and 1972 to randomly collect the validation data from the raw images (Biro et al., 2013; Lung & Schaab, 2010; Waiswa, 2011). Background knowledge of the study area as well as qualitative information from local informants were also helpful in the selection of these validation samples sites.

A confusion matrix, the overall accuracy, and the Kappa index of agreement were reported for each LUC maps (Appendix 9). Ultimately, some post-classification analyses were performed to minimize classification errors due to image registration and georeferencing of satellite images. A clump of 3 x 3 window was applied to all output maps to eliminate the “salt and pepper” polygons (Petursson et al., 2013; Zhai et al., 2015).

b) Analyses of the change and patterns of LUC types

The output images were exported to GIS software for change detection analyses between the four individual maps of the basin. The post-classification comparison adopted to detect changes in land-cover types was based on pairwise overlay (bi-temporal analyses) of individual LUC maps (Badjana et al., 2014; Braimoh & Vlek, 2004b; Pang et al., 2010; Pang et al., 2013). Class statistics, transitions analyses, conversion categories, and annual rates of occurrence were computed from the output LUC maps for each LUC type and transition category (Tfwala et al., 2012; White et al., 2013; Zhou et al., 2008). Statistics were produced for the three transitional periods 1972 - 1987; 1987 - 2000; 2000 - 2014 and the overall period 1972 - 2014.

The gross gains (total gains), gross losses (total losses), net change (i.e. changes in land quantity), and swap changes (i.e. changes in land location) were detected for each LUC type from a pairwise conversion matrix (Braimoh, 2004; Carmona & Nahuelhual, 2012; Pontius et al., 2004; Schmitt-Harsh, 2013). The gross gain for a category *i* was expressed as the summative value of all areas gained from other LUC types at a final date. Inversely, the gross loss of a type *i* was the summative value of all areas converted from *i* into other LUC types. For each LUC type, the total change area was calculated as the sum of all areas affected by changes, (i.e. gross gains + gross losses). The net change was calculated as the simple difference between gross gains and losses during a transition period. Meanwhile, the swap change was derived by subtracting the absolute net change from the total change for the specific cover type during a given transition period.

Net annual change for each LUC type was calculated according to Equation 5.1 below (Carmona & Nahuelhual, 2012; FAO, 1996):

$$C_R = \ln\left[\frac{Sf}{Si}\right] \times \frac{100}{Tf - Ti} \qquad \text{(Equation 5.1)}$$

where C_R is the annual rate of change of a LUC type, *Sf* is the area of the targeted LUC type at the final time *Tf*; *Si* is the area of the same targeted LUC type at the initial time *Ti*; ln is the natural log function.

Furthermore, the important land cover changes and persistence were detected among the different cover types for all the transition periods following the method of Pontius et al. (2004). This methodology assumes there is randomness in the landscape transitions when land categories gained from other categories in proportion to the availability of the other losing categories, or reciprocally. Meanwhile the systematic transitions were analysed by interpreting the transition proportions relative to the sizes of the categories. A transition is assumed random when the difference between the expected and the actual transition proportions is close to zero while any large value indicates systematic landscape transition (Pontius et al., 2004; Romero-Ruiz et al., 2012; Schmitt-Harsh, 2013). For this study, transitions with an absolute difference value higher than or equal to 0.5 were considered as the most important systematic changes.

In the process of computing the proportions accounting for the systematic and random changes, three important variables were used viz. the observed (actual) transition values, the expected land gains and losses under random processes of gain and loss (Gutiérrez Angonese & Grau, 2014; Romero-Ruiz et al., 2012; Teferi et al., 2013). Whilst the observed transitions were computed from the actual values in the cross-tabulation matrix between two times, Equations 5.2 and 5.3 were used to calculate the expected gain (G_{ij}) and expected loss (L_{ij}) of each transition under a random process of gain or loss (Nakakaawa et al., 2010; Pontius et al., 2004; Schmitt-Harsh, 2013).

$$G_{ij} = \left(P_{+j} - P_{jj}\right)\left(\frac{P_{i+}}{\sum_{i=1} P_{i+}}\right), \ \forall\ i \neq j \qquad \text{(Equation 5.2)}$$

$$L_{ij} = \left(P_{i+} - P_{ii}\right)\left(\frac{P_{+j}}{\sum_{j=1} P_{+j}}\right), \ \forall\ i \neq j \qquad \text{(Equation 5.3)}$$

where G_{ij} denotes the expected transition from category i to j under random processes of gain, P_{+j} is the proportion of the landscape in category j in the final time; P_{jj} is the observed persistent proportion of the category j; P_{i+} is the total area of category i at initial time; L_{ij} is the expected transition from category i to j under random processes of loss; and P_{ii} is the persistence proportion of the category i between the two times.

In addition, loss-to-persistence ratio $L_{(-)}$ and gain-to-persistence ratio $G_{(+)}$were also calculated using Equations 5.4 and 5.5 to assess the vulnerability of the LUC types to transition (Braimoh, 2004; Nakakaawa et al., 2010; Ouedraogo, 2010; Romero-Ruiz et al., 2012). $L_{(-)}$ value for a cover category higher than 1 indicates a high vulnerability of that LUC types to be converted into other categories (Gutiérrez Angonese & Grau, 2014). $G_{(+)}$ indicates the tendency of a cover category to gain more from other cover types. These ratios were expressed as follows:

$$L_{(-)} = \left(\frac{\text{Observed loss - expected loss}}{\text{Expected loss}}\right), \text{ in \%} \qquad \text{(Equation 5.4)}$$

$$G_{(+)} = \left(\frac{\text{Observed gain - expected gain}}{\text{Expected gain}}\right), \text{ in \%} \qquad \text{(Equation 5.5)}$$

where the expected gains and losses are *Gij* and *Lij* from Equations 5.2 and 5.3, respectively. The observed gain and loss are the actual values from the transition analyses.

5.3. Results

5.3.1. Historical LUC types and spatial patterns

During the last four decades, major transformations affected the landscapes of the Mo river basin (Figure 5.2). Spatially, the central and the northeast areas of the basin were dominantly covered by human systems (croplands and settlements), especially for the 2000 and 2014. Most of the greenest areas from 1972 to 2014 lie within protected areas, especially the Fazao-Malfakassa National Park in the southern and western parts of the basin. Savannahs and shrubs were the most widely scattered LUC types from the earlier dates to the most recent years. Some scattered patches of forests and woodlands located within the most human-dominated parts, decreased markedly from 1972 to 2014. Meanwhile, the emergence of forest class was marked along river network within the protected areas whereas in the free access lands, their cover decreased consistently over time. Built up areas were mostly developed in the eastern parts along the main roads.

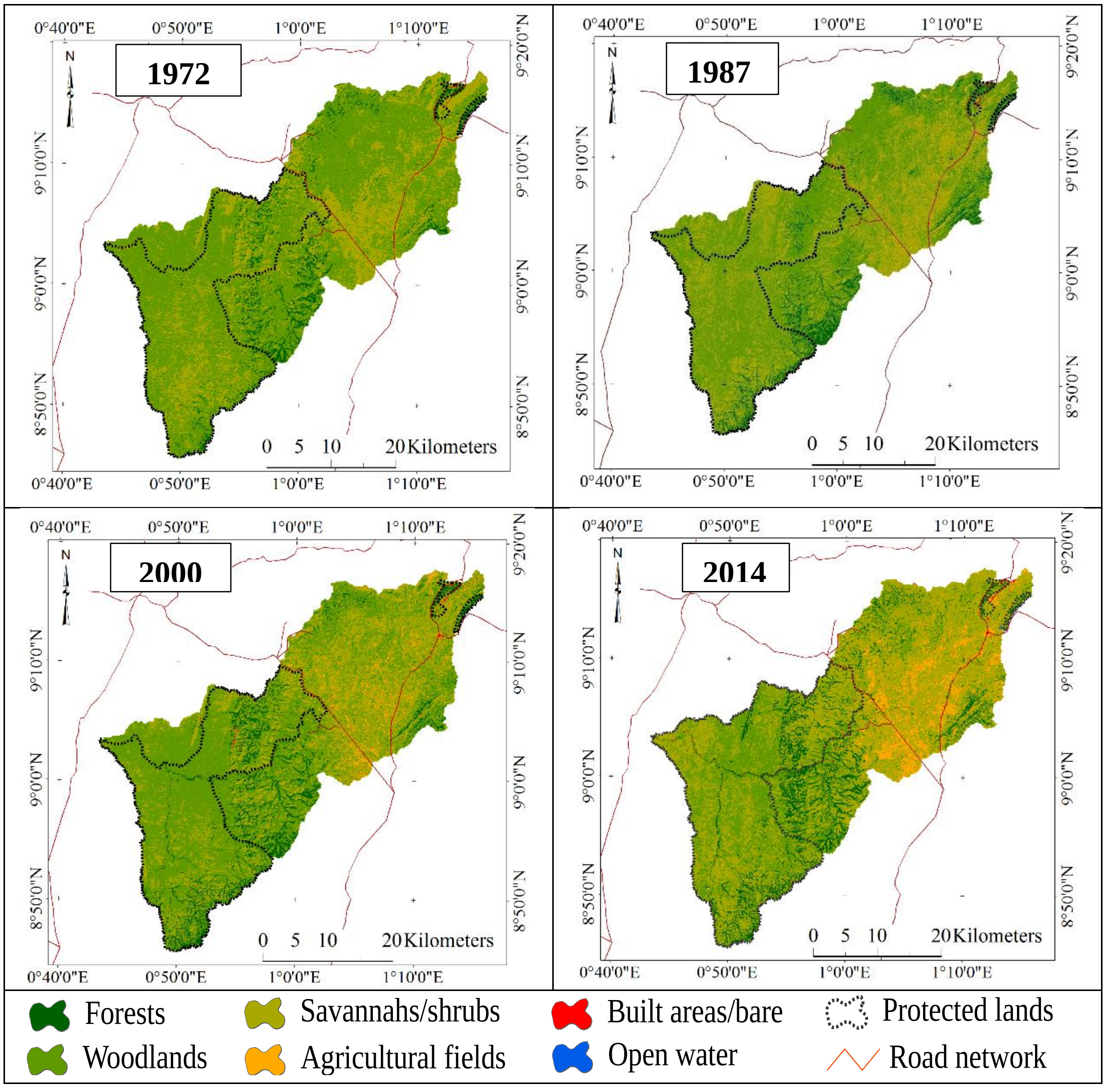

Figure 5.2. Historical LUC types for the Mo river basin

In terms of the areal distribution, it is showed that natural vegetation dominated the landscapes and decreased from 99 % in 1972 to 98 %, 96 % and 91 %, respectively in 1987, 2000 and 2014 (Tables 5.2a, b, c and d). The most dominant cover types over time were woodlands and savannahs. While the total area of the woodlands decreased from 94,829 ha in 1972 to 40,527 ha in 2014, savannahs increased from 45,000 ha (30 %) in 1972 to about 79,000 ha (53 %) in 2014. The proportion of forests was about 6 % of the total area in 1972 and slightly increased to 6 %, 9 % and 11 % for 1987, 2000 and 2014, respectively. Over the period 1972-2014, there was an increasing forest coverage. Croplands significantly increased from 0.2 % in 1972 to 8 % in 2014. Similarly, built areas also showed an increasing trend all over the years but still exhibited very low values. In general, All LUC types experienced unidirectional changes between 1972 and 2014, with the exception of water areas, which varied in relation to the water levels in Aleheride dams and riverbeds. The class statistics indicated high wildness of the study area where agricultural and settlement expansion are increasingly reducing the landscape wilderness.

5.3.2. Change rates, persistence, gains and losses of LUC types

Figure 5.3 gives an overview of the historical LUCC in the Mo basin. The detailed pairwise transition matrices for the four periods are provided in Appendix 10. Forests gained about 791 ha, 3497 ha and 3712 ha, respectively for the three transition periods. Though forests gained over all the periods, the annual rates of gain decreased during the second and third transition periods (Tables 5.2b and 5.2c).

The swap changes for forest cover were of 7.2 %; 6.9 % and 5.9 % for the 3 transition periods, respectively. The swap values were quite higher than the respective absolute net changes (0.5 %; 2.4 % and 2.5 %), suggesting that forest regeneration (gain from other LUC types) and degradation or deforestation (loss to other LUC types) during all the transition periods affected much more spatial extent than revealed through the net changes. However, the overall net (5.4 %) and swap (6.1 %) changes over the period 1972-2014 were closer, indicating that forest changes (loss and gain) affected mostly the greatest proportions of its initial coverage in addition to the net gain of about 0.7 % of new forests.

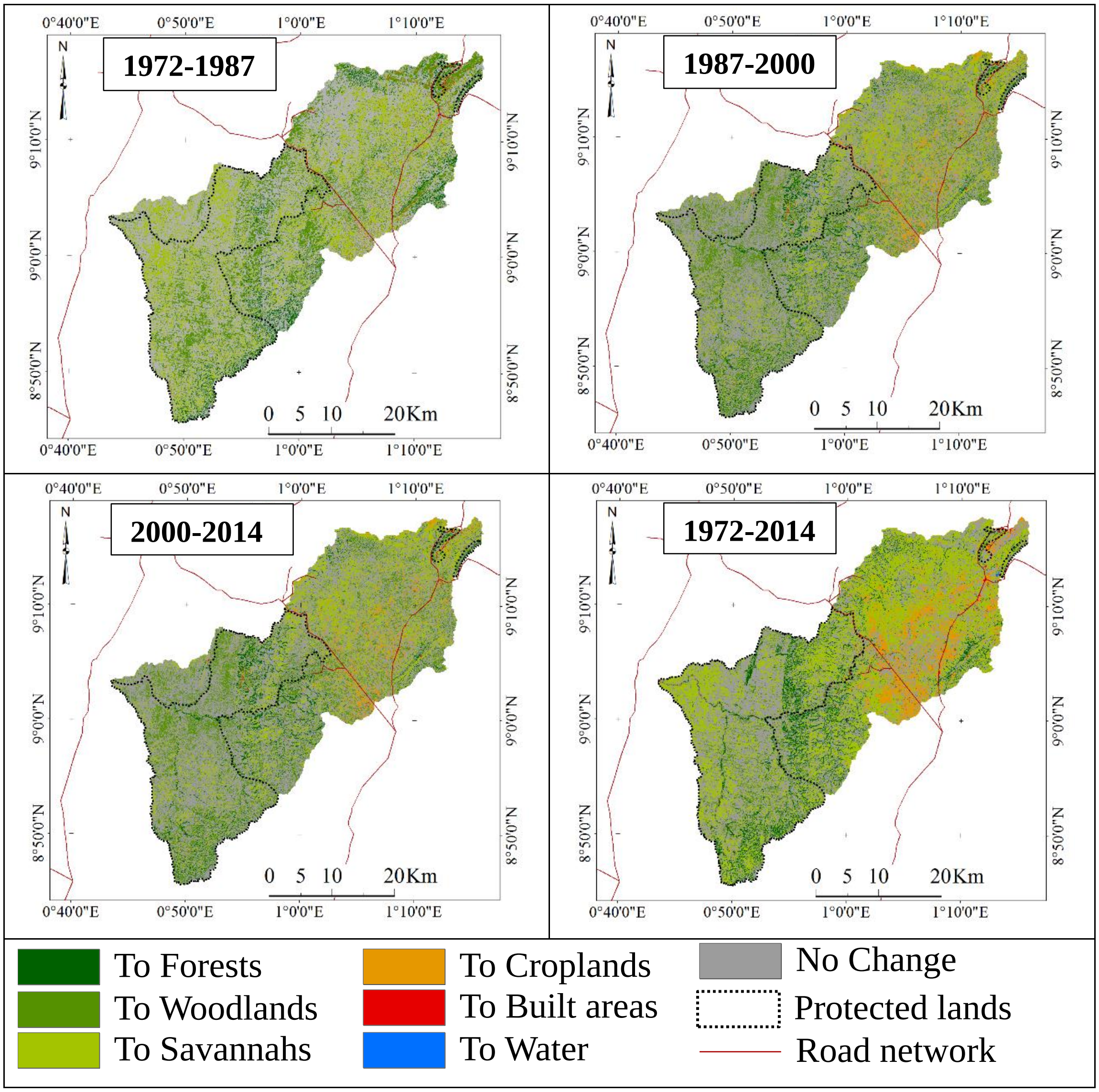

Figure 5.3. LUC transition maps for the four transition periods

Similar to forests, savannahs showed a constant increase in coverage from a net gain of 3,051 ha during the period 1 to an approximate net gain of 1,876 ha and 29,028 ha during the periods 2 and 3, respectively (Tables 5.1). However, land transformation affected more the coverage (high swap values of 34.6 %, 34.7 % and 21 % for periods 1, 2, and 3, respectively). Period 2 showed the lowest annual rate of gain (0.3 %) while period 3 experienced the highest processes of savannah gain (3.3 %). The overall net (23 %) and swap (25 %) changes showed that the processes of savannah gain and loss affected similar landscapes, although an additional 2 % of savannah cover change was not captured by the net change. During the overall period, this category has the highest swap (25 %), suggesting its constant transformation (losses to and gains from other categories).

In contrast, woodlands experienced net losses of about 3.9 %, 5.4 % and 27.3 % of their initial coverage during the 3 periods, respectively (Tables 5.1). Similar to the savannahs, woodlands experienced a high swap change of 38.2 % and 35.6 % during the first two periods, respectively. This indicated that the woodlands experienced high exchanges of cover location (gain and loss) with other LUC types. The swap in woodlands was low for the third period (12.3 %) because this category considerably decreased over time. Between 1972 and 2014, woodlands had a net loss of about 36.7 % out of 49.3 % of total change at an annual rate of - 2 %. The highest rate of woodland loss occurred during the period 1987 - 2000. During the overall time, the swap for woodlands represented about 26 % of the total change, which is lower than the decreasing swap changes for this category during transition periods (91 %, 87 %, and 31 % for the 3 periods, respectively).

During the three transitional periods, transformations affecting cultivated land consisted of both swap and net changes (Tables 5.1). While net gains of croplands increased over time, the swapping affected additional areas passing from 0.3 % between 1972 and 1987 to 2.4 % and 3.6 % for the transition periods 2 and 3, respectively. Similarly, swap of settlements was greater than the net changes, meaning that settlement expansion was underestimated by the net change. Thus, cultivated lands and settlements showed high tendency to gain from other categories much more than losing, suggesting that settlement expansion as well as intensive and extensive cultivation occurred in the Mo basin.

Tables 5.1. Areal distribution of the four main LUC types and changes

a. Changes for the period 1 (1972 – 1987)

1972-1987	Total 1972		Total 1987		Total gains		Total losses		Total change		Net change		Swap change		Annual change
	Areas		Areas		Areas		Areas		Areas		Areas		Areas		Rate
LUC types	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%	%
Forests	8367.48	5.64	9152.28	6.17	6126.03	4.13	5334.75	3.60	11460.78	7.73	+ 791.28	+ 0.53	10669.50	7.20	0.60
Woodlands	94643.01	63.85	88993.53	59.99	28302.39	19.09	34040.79	22.96	62343.18	42.06	- 5738.40	- 3.87	56604.78	38.19	- 0.41
Savannahs	44891.64	30.28	47983.86	32.34	28713.33	19.37	25662.33	17.31	54375.66	36.68	+ 3051.00	+ 2.06	51324.66	34.62	0.44
Croplands	258.21	0.17	2024.37	1.36	2010.69	1.36	243.36	0.16	2254.05	1.52	+ 1767.33	+ 1.19	486.72	0.33	13.73
Settlements	54.27	0.04	173.07	0.12	169.20	0.11	50.22	0.03	219.42	0.15	+ 118.98	+ 0.08	100.44	0.07	7.75
Water	16.47	0.01	26.28	0.02	26.28	0.02	16.47	0.01	42.75	0.03	+ 9.81	+ 0.01	32.94	0.02	2.97

b. Changes for the period 2 (1987 - 2000)

1987-2000	Total 1987		Total 2000		Total gains		Total losses		Total change		Net change		Swap change		Annual change
	Areas		Areas		Areas		Areas		Areas		Areas		Areas		Rate
	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%	%
Forests	9152.28	6.17	12647.97	8.53	7870.05	5.30	4372.74	2.95	12242.79	8.25	+ 3497.31	+ 2.36	8745.48	5.90	2.46
Woodlands	88993.53	59.99	80993.16	54.60	26395.47	17.79	34382.88	23.18	60778.35	40.97	- 7987.41	- 5.38	52790.94	35.58	- 0.73
Savannahs	47983.86	32.34	49850.64	33.61	27587.79	18.60	25712.01	17.33	53299.80	35.93	+ 1875.78	+ 1.26	51424.02	34.66	0.30
Croplands	2024.37	1.36	4471.38	3.01	4247.28	2.86	1799.19	1.21	6046.47	4.08	+ 2448.09	+ 1.65	3598.38	2.43	6.08
Settlements	173.07	0.12	363.69	0.25	341.10	0.23	150.30	0.10	491.40	0.33	+ 190.80	+ 0.13	300.60	0.20	5.73
Water	26.28	0.02	1.71	0.00	0.09	0.00	24.66	0.02	24.75	0.02	- 24.57	- 0.02	0.18	0.00	- 21.02

c. Changes for the period 3 (2000 - 2014)

2000-2014	Total 2000		Total 2014		Total gains		Total losses		Total change		Net change		Swap change		Annual change
	Areas		Areas		Areas		Areas		Areas		Areas		Areas		Rate
	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%	%
Forests	12647.97	8.53	16350.03	11.03	8827.92	5.95	5115.60	3.45	13943.52	9.40	+ 3712.32	+ 2.50	10231.20	6.90	1.84
Woodlands	80993.16	54.60	40448.16	27.29	9153.99	6.17	49673.16	33.49	58827.15	39.66	- 40519.17	- 27.32	18307.98	12.34	- 4.95
Savannahs	49850.64	33.61	78818.31	53.17	44604.18	30.07	15576.48	10.50	60180.66	40.57	+ 29027.70	+ 19.57	31152.96	21.00	3.28
Croplands	4471.38	3.01	12166.38	8.21	10358.46	6.98	2669.67	1.80	13028.13	8.78	+ 7688.79	+ 5.18	5339.34	3.60	7.15
Settlements	363.69	0.25	426.69	0.29	375.30	0.25	311.94	0.21	687.24	0.46	+ 63.36	+ 0.04	623.88	0.42	1.11
Water	1.71	0.00	28.71	0.02	27.18	0.02	0.18	0.00	27.36	0.02	+ 27.00	+ 0.02	0.36	0.00	20.15

d. Changes for the overall period (1972 - 2014)

1972-2014	Total 1972		Total 2014		Total gains		Total losses		Total change		Net change		Swap change		Annual change
	Areas		Areas		Areas		Areas		Areas		Areas		Areas		Rate
	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%	%
Forests	8367.48	5.64	16350.03	11.03	12507.39	8.44	4526.73	3.05	17034.12	11.49	+ 7980.66	+ 5.38	9053.46	6.11	1.59
Woodlands	94643.01	63.85	40448.16	27.29	9421.02	6.36	63619.74	42.92	73040.76	49.27	- 54198.72	- 36.56	18842.04	12.71	- 2.02
Savannahs	44891.64	30.28	78818.31	53.17	52536.78	35.44	18611.28	12.55	71148.06	48.00	+ 33925.50	+ 22.89	37222.56	25.11	1.34
Croplands	258.21	0.17	12166.38	8.21	12066.93	8.14	158.94	0.11	12225.87	8.25	+ 11907.99	+ 8.03	317.88	0.21	9.16
Settlements	54.27	0.04	426.69	0.29	420.21	0.28	47.79	0.03	468.00	0.32	+ 372.42	+ 0.25	95.58	0.06	4.91
Water	16.47	0.01	28.71	0.02	28.44	0.02	16.29	0.01	44.73	0.03	+ 12.15	+ 0.01	32.58	0.02	1.27

5.3.3. Land use cover persistence, systematic and random conversions

Land cover persistence dominated the landscape (more than 50 % as sum of first lines of diagonal values in Tables 5.2). The persistence constantly decreased and accounted for 56 %, 55 %, and 50 % of the landscape, respectively for the 3 transition periods. About 59 % (100 minus the sum of the diagonal entries in first lines in Tables 5.3) of the landscape did change during the overall 42-year period. Natural vegetation dominated the persistence with woodlands exhibiting the highest but decreasing persistence over time. Similarly, the persistence of forests and savannahs constantly increased for all the transition periods. These latter categories gained much more from the transformation of woodlands.

Difference in values between the observed and expected proportions of the landscapes (values in round parentheses in all Tables 5.2) are used to detect random or systematic transitions. Values closer to zero are indicative of random transition while higher values indicate systematic transitions (but a threshold of 0.5 % was considered for analytical purpose in this study). Most of the major transitions occurred between the four dominant cover categories, viz. forests, woodlands, savannahs and croplands. Over the three and the overall periods, transitions forests-savannahs indicated large and negative difference values between observed and expected gains for savannahs, ranging from -1.18 % (1972 - 1987) to -2.52 % (2000 - 2014). These negative values indicated that savannahs did not emerge from forests. Similarly, a negative difference value (-1.12 %) was observed for the overall period 1972 - 2014, corroborating the general trend of systematic avoidance of forest replacement under random gain processes for savannahs. The vulnerability of forests to loss was evidenced by $L_{(-)}$ values higher than 1 during the first transition and overall periods (Tables 5.2a & d). Similarly, $G_{(+)}$ for forests and savannahs were higher than 1, indicating that this categories gained much more than persistence over time. In contrast, $L_{(-)}$ of woodlands were of 1.6 for 2000 - 2014 and 2.1 for 1972 - 2014 indicating their high vulnerability to loss in recent years in comparison with the two first periods where woodlands were less vulnerable ($L_{(-)} < 1$).

Conversely, forest gains were not associated with the replacement of savannahs (negative values of difference between observed and expected gains; -0.74 %; -1.31 %; -0.44 % and -0.18 %). In term of gains, there is a systematic mutual avoidance between forests and savannahs, in line with the trend of the transition forests-savannahs. However, forests systematically gained from woodlands (0.74 %, 1.38 %, and 0.57 %) rather than savannahs, which exhibited negative values of observed minus expected gains of forests from savannahs. Mostly, savannah gains during the four periods emerged from the replacement of woodlands, as indicated by the positive values of difference observed-expected gains (1.15 %; 0.76 %, 2.30 % and 1.12 %, respectively for the four periods). Under expected random gain, Croplands systematically emerged from savannah losses solely at the rates varying between 0.8-1. In overall, under random process of gain, Forests gained more from Woodlands than the inverse at relatively very low expected rates (ratio values close to zero). Forests gains during all the periods did not emerge from Savannahs, and vice versa. Savannahs mostly gained from woodlands rather than from forests while woodlands did not gain from savannahs, except during the period 1987-2000.

On the other hand, as indicated by the positive values of difference between observed and expected losses, forest losses during the three transition periods occurred systematically towards woodlands (0.92 %; 0.76 %, and 0.98 %, for period 1, 2, and 3, respectively) rather than savannahs (-0.88 %, -0.66 %, -0.72 %, respectively for the periods 1, 2 and 3). However, in line with the criteria of systematic change defined in this study (threshold of 0.5 %), the overall period did not show a systematic gain of woodlands from forests (0.24 % lower than 0.5 %). Under these random processes of loss, it was expected that the loss of savannahs would be systematically converted into woodlands, except the periods 2000-2014 (-2.11 %) and 1972-2014 (-2.15 %), indicating that savannah losses are not associated with woodland replacement for these latter transition periods. The transition croplands-natural vegetation showed that the loss of croplands tended to be due to the systematic conversion into savannahs rather than other natural categories. This trend was most acute for the period 2000-2014 with a difference in value between observed and expected loss of 0.57 % and at an expected ratio around 0.5. Neither savannahs nor croplands were systematically converted into forests, as indicated by the low proportions of transition. Croplands exhibited highest values of $G_{(+)}$ indicating the agricultural expansion occurred much more than agricultural land abandonment or conversion into other cover categories. These $G_{(+)}$ values for croplands were far higher than $L_{(-)}$ values indicating that croplands are less vulnerable to conversion to other categories than they gain from other categories.

Tables 5.2: Matrices for the four periods under investigation in the Mo River basin

a: Period 1 (1972-1987)

		1987													
		Forests		Woodlands		Savannahs		Croplands		Settlements		Water		Total 1972	Gross loss
1972	Forests	**2.05**	Ratio	**3.22**	Ratio	**0.36**	Ratio	**0.01**	Ratio	**0.00**	Ratio	**0.00**	Ratio	**5.64**	**3.60**
		2.05 (0.00)	0.00	2.98 (0.24)	0.08	1.55 (-1.18)	-0.76	0.08 (-0.07)	-0.87	0.01 (-0.01)	-0.91	0.00 (0.00)	-0.88	6.66 (-1.02)	4.61 (-1.01)
		_2.05 (0.00)_	_0.00_	_2.30 (0.92)_*	_0.40_	_1.24 (-0.88)_*	_-0.71_	_0.05 (-0.04)_	_-0.81_	_0.00 (0.00)_	_-0.86_	_0.00 (0.00)_	_-0.82_	_5.65 (0.00)_	_3.60 (0.00)_
	Woodlands	**3.54**		**40.88**		**18.89**	0.00	**0.50**	0.00	**0.02**	0.00	**0.01**	0.00	**63.85**	**22.96**
		2.80 (0.74)	0.27	40.88 (0.00)	0.00	17.74 (1.15)	0.06	0.87 (-0.37)	-0.42	0.07 (-0.05)	-0.67	0.01 (-0.01)	-0.54	62.37 (1.48)	21.49 (1.48)
		_3.55 (0.00)_	_0.00_	_40.88 (0.00)_	_0.00_	_18.56 (0.34)_	_0.02_	_0.78 (-0.28)_	_-0.36_	_0.07 (-0.04)_	_-0.64_	_0.01 (-0.01)_	_-0.49_	_63.85 (0.00)_	_22.96 (0.00)_
	Savannahs	**0.59**		**15.78**		**12.97**	0.00	**0.84**	0.00	**0.09**	0.00	**0.01**	0.00	**30.28**	**17.31**
		1.33 (-0.74)	-0.56	15.99 (-0.21)	-0.01	12.97 (0.00)	0.00	0.41 (0.43)	1.05	0.03 (0.05)	1.47	0.01 (0.01)	1.17	30.75 (-0.46)	17.77 (-0.46)
		_1.58 (-0.99)_*	_-0.63_	_15.35 (0.44)_	_0.03_	_12.97 (0.00)_	_0.00_	_0.35 (0.49)_	_1.41_	_0.03 (0.06)_	_1.85_	_0.00 (0.01)_	_1.57_	_30.28 (0.00)_	_17.31 (0.00)_
	Croplands	**0.00**		**0.07**		**0.09**	0.00	**0.01**	0.00	**0.00**	0.00	**0.00**	0.00	**0.17**	**0.16**
		0.01 (-0.01)	-0.85	0.09 (-0.02)	-0.24	0.05 (0.04)	0.82	0.01 (0.00)	0.00	0.00 (0.00)	572.86	0.00 (0.00)	24.56	0.16 (0.02)	0.15 (0.02)
		_0.01 (-0.01)_	_-0.89_	_0.10 (-0.03)_	_-0.30_	_0.05 (0.03)_	_0.64_	_0.01 (0.00)_	_0.00_	_0.00 (0.00)_	_18.66_	_0.00 (0.00)_	_25.75_	_0.17 (0.00)_	_0.16 (0.00)_
	Settlements	**0.00**		**0.01**		**0.02**		**0.00**	0.00	**0.00**	0.00	**0.00**		0.04	0.03
		0.00 (0.00)	-0.77	0.02 (-0.01)	-0.40	0.01 (0.01)	0.83	0.00 (0.00)	5.83	0.00 (0.00)	0.00	0.00 (0.00)	-1.00	0.03 (0.00)	0.03 (0.00)
		_0.00 (0.00)_	_-0.83_	_0.02 (-0.01)_	_-0.43_	_0.01 (0.01)_	_0.69_	_0.00 (0.00)_	_6.34_	_0.00 (0.00)_	_0.00_	_0.00 (0.00)_	_-1.00_	_0.04 (0.00)_	_0.03 (0.00)_
	Water	**0.00**		**0.01**		**0.01**		**0.00**	0.00	**0.00**	0.00	**0.00**		0.01	0.01
		0.00 (0.00)	-1.00	0.01 (0.00)	-0.14	0.00 (0.00)	0.93	0.00 (0.00)	-0.60	0.00 (0.00)	3.79	0.00 (0.00)	0.00	0.01 (0.00)	0.01 (0.00)
		_0.00 (0.00)_	_-1.00_	_0.01 (0.00)_	_-0.24_	_0.00 (0.00)_	_0.66_	_0.00 (0.00)_	_-0.60_	_0.00 (0.00)_	_3.67_	_0.00 (0.00)_	_0.00_	_0.01 (0.00)_	_0.01 (0.00)_
	Total 1987	**6.18**		**59.98**		**32.34**		**1.37**	0.00	**0.12**	0.00	**0.02**	0.00	100.00	44.09
		6.18	0.00	59.98 (0.00)	0.00	32.34 (0.00)	0.00	1.37 (0.00)	0.00	0.12 (0.00)	0.00	0.02 (0.00)	0.00	100.00 (0.00)	44.06
		_7.19 (-1.01)_	_-0.14_	_58.66 (1.32)_	_0.02_	_32.84 (-0.50)_	_-0.02_	_1.20 (0.17)_	_0.14_	_0.10 (0.01)_	_0.12_	_0.02 (0.00)_	_0.15_	_100 (0.00)_	_44.09_
	Gross Gain	**4.13**		**19.09**		**19.37**		**1.36**		**0.11**		**0.02**		44.09	
		4.13 (0.00)		19.09 (0.00)		19.37 (0.00)		1.36 (0.00)		0.11 (0.00)	0.00	0.02 (0.00)		44.08	
		_5.14 (-1.01)_		_17.77 (1.32)_		_19.87 (-0.50)_		_1.19 (0.17)_		_0.10 (0.01)_	_0.00_	_0.02 (0.00)_		_44.08_	

Note: Each cell is subdivided into three rows and two columns of numbers. Left column of each cell: the first row contains bolded numbers that represent the actual (observed) proportions of inter-categorical transitions (persistence and transitions) of the landscape. The second row represents the expected percentage of land under random processes of gain (named Expected (+)) calculated using Equation 2, where figures in round parentheses are equal to the observed proportion minus the one expected (named Difference (+)). The third row contains italicized numbers representing the expected proportion of land under random processes of loss (named Expected (-)) calculated using Equation 3, where numbers within round parentheses represent the observed proportion minus the expected one (named Difference (-)). Extreme right column of the table contains the Loss-to-persistence ratio ($L_{(-)}$) while the extreme row is the Gain-to-persistence ratio ($G_{(+)}$). Numbers highlighted in gray represent systematic gain transitions; starred numbers are the systematic loss transitions.

b: Period 2 (1987-2000)

		2000													
		Forests		Woodlands		Savannahs		Croplands		Settlements		Water		Total 1987	Gross loss
1987	Forests	**3.22**	Ratio	**2.52**	Ratio	**0.42**	Ratio	**0.01**	Ratio	**0.00**	Ratio	**0.00**	Ratio	**6.17**	**2.95**
		3.22 (0.00)	0.00	2.74 (-0.23)	-0.08	1.70 (-1.28)	-0.75	0.18 (-0.17)	-0.94	0.01 (-0.01)	-0.97	0.00 (0.00)	-1.00	7.85 (-1.68)	4.63 (-1.68)
		3.22 (0.00)	*0.00*	*1.76 (0.76)**	*0.43*	*1.08 (-0.66)**	*-0.61*	*0.10 (-0.09)*	*-0.89*	*0.01 (-0.01)*	*-0.95*	*0.00 (0.00)*	*-1.00*	*6.17 (0.00)*	*2.95 (0.00)*
	Woodlands	**4.77**		**36.81**		**17.25**		**1.09**		**0.07**		**0.00**		**59.99**	**23.18**
		3.39 (1.38)	0.41	36.81 (0.00)	0.00	16.49 (0.76)	0.05	1.74 (-0.65)	-0.37	0.14 (-0.07)	-0.51	0.00 (0.00	0.67	58.57 (1.42)	21.76 (1.42)
		4.35 (0.42)	*0.10*	*36.81 (0.00)*	*0.00*	*17.16 (0.09)*	*0.01*	*1.54 (-0.45)*	*-0.29*	*0.13 (-0.06)*	*-0.46*	*0.00 (0.00)*	*-0.90*	*59.99 (0.00)*	*23.18 (0.00)*
	Savannahs	**0.52**		**14.92**		**15.01**		**1.74**		**0.15**		**0.00**		**32.34**	**17.33**
		1.83 (-1.31)	-0.71	14.38 (0.54)	0.04	15.01 (0.00)	0.00	0.94 (0.80)	0.85	0.07 (0.07)	0.96	0.00 (0.00)	-1.00	32.24 (0.11)	17.22 (0.11)
		*2.23 (-1.70)**	*-0.77*	*14.25 (0.67)**	*0.05*	*15.01 (0.00)*	*0.00*	*0.79 (0.95)**	*1.21*	*0.06 (0.08)*	*1.28*	*0.00 (0.00)*	*-1.00*	*32.34 (0.00)*	*17.33 (0.00)*
	Croplands	**0.01**		**0.33**		**0.85**		**0.15**		**0.02**		**0.00**		**1.36**	**1.21**
		0.08 (-0.07)	-0.85	0.61 (-0.27)	-0.45	0.38 (0.48)	1.28	0.15 (0.00)	0.00	0.00 (0.01)	3.85	0.00 (0.00)	-1.00	1.21 (0.15)	1.06 (0.15)
		0.11 (-0.10)	*-0.89*	*0.68 (-0.35)*	*-0.51*	*0.42 (0.43)*	*1.03*	*0.15 (0.00)*	*0.00*	*0.00 (0.01)*	*3.96*	*0.00 (0.00)*	*-1.00*	*1.36 (0.00)*	*1.21 (0.00)*
	Settlements	**0.00**		**0.02**		**0.06**		**0.02**		**0.02**		**0.00**		**0.12**	**0.10**
		0.01 (-0.01)	-0.90	0.05 (-0.04)	-0.71	0.03 (0.03)	0.93	0.00 (0.02)	5.90	0.02 (0.00)	0.00	0.00 (0.00)	-1.00	0.11 (0.01)	0.09 (0.01)
		0.01 (-0.01)	*-0.92*	*0.06 (-0.04)*	*-0.72*	*0.03 (0.03)*	*0.82*	*0.00 (0.02)*	*6.63*	*0.02 (0.00)*	*0.00*	*0.00 (0.00)*	*-1.00*	*0.12 (0.00)*	*0.10 (0.00)*
	Water	**0.00**		**0.01**		**0.01**		**0.00**		**0.00**		**0.00**		**0.02**	**0.02**
		0.00 (0.00)	1.06	0.01 (0.00)	-0.34	0.00 (0.00)	0.78	0.00 (0.00)	-0.41	0.00 (0.00)	7.93	0.00 (0.00)	0.00	0.02 (0.00)	0.01 (0.00)
		0.00 (0.00)	*0.46*	*0.01 (0.00)*	*-0.43*	*0.01 (0.00)*	*0.55*	*0.00 (0.00)*	*-0.39*	*0.00 (0.00)*	*7.93*	*0.00 (0.00)*	*0.00*	*0.02 (0.00)*	*0.02 (0.00)*
	Total 2000	**8.53**		**54.60**		**33.61**		**3.01**		**0.25**		**0.00**		**100.00**	44.79
		8.53 (0.00)		54.60 (0.00)		33.61 (0.00)		3.01 (0.00)		0.25 (0.00)		0.00 (0.00)		100.00 (0.00)	44.79
		9.92 (-1.39)		*53.57 (1.03)*		*33.71 (-0.11)*		*2.58 (0.44)*		*0.22 (0.03)*		*0.00 (0.00)*		*100.00*	
	Gross Gain	**5.30**		**17.79**		**18.60**		**2.86**		**0.23**		**0.00**		44.79	
		5.30 (0.00)		17.79 (0.00)		18.60 (0.00)		2.86 (0.00)		0.23 (0.00)		0.00 (0.00)		44.79	
		6.70 (-1.39)		*16.76 (1.03)*		*18.70 (-0.11)*		*2.43 (0.44)*		*0.20 (0.03)*		*0.00 (0.00)*		*0.00*	

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c: Period 3 (2000 - 2014)

		2014														
		Forests		Woodlands		Savannahs		Croplands		Settlements		Water		Total 2000	Gross loss	
2000	Forests	**5.08**		**2.04**		**1.34**		**0.06**		**0.01**		**0.00**		**8.53**	**3.45**	
		5.08 (0.00)	0.00	1.16 (0.88)	0.76	3.86 (-2.52)	-0.65	0.61 (-0.55)	-0.90	0.02 (-0.02)	-0.75	0.00 (0.00)	-0.38	10.74 (-2.21)	5.66	-2.21
		5.08 (0.00)	*0.00*	*1.06 (0.98)**	*0.93*	*2.06 (-0.72)**	*-0.35*	*0.32 (-0.26)*	*-0.81*	*0.01 (-0.01)*	*-0.52*	*0.00 (0.00)*	*0.29*	*8.53 (0.00)*	*3.45*	*0.00*
	Woodlands	**4.12**		**21.12**		**27.03**		**2.26**		**0.07**		**0.01**		**54.60**	**33.49**	
		3.55 (0.57)	0.16	21.12 (0.00)	0.00	24.73 (2.30)	0.09	3.93 (-1.67)	-0.42	0.14 (-0.07)	-0.53	0.01 (0.00)	-0.16	53.48 (1.12)	32.37	1.12
		*5.08 (-0.96)**	*-0.19*	*21.12 (0.00)*	*0.00*	*24.49 (2.54)**	*0.10*	*3.78 (-1.51)**	*-0.40*	*0.13 (-0.07)*	*-0.51*	*0.01 (0.00)*	*-0.05*	*54.60 (0.00)*	*33.49*	*0.00*
	Savannahs	**1.75**		**4.01**		**23.11**		**4.58**		**0.16**		**0.01**		**33.61**	**10.50**	
		2.19 (-0.44)	-0.20	4.57 (-0.56)	-0.12	23.11 (0.00)	0.00	2.42 (2.16)	0.89	0.09 (0.07)	0.84	0.01 (0.00)	0.38	32.37 (1.23)	9.27	1.23
		*2.47 (-0.72)**	*-0.29*	*6.12 (-2.11)**	*-0.34*	*23.11 (0.00)*	*0.00*	*1.84 (2.74)**	*1.49*	*0.06 (0.09)*	*1.43*	*0.00 (0.00)*	*0.96*	*33.61 (0.00)*	*10.50*	*0.00*
	Croplands	**0.05**		**0.11**		**1.61**		**1.21**		**0.03**		**0.00**		**3.01**	**1.80**	
		0.20 (-0.14)	-0.73	0.41 (-0.30)	-0.72	1.37 (0.24)	0.18	1.21 (0.00)	0.00	0.01 (0.02)	2.33	0.00 (0.00)	-0.45	3.19 (-0.18)	1.98	-0.18
		0.22 (-0.16)	*-0.76*	*0.53 (-0.42)*	*-0.79*	*1.04 (0.57)**	*0.54*	*1.21 (0.00)*	*0.00*	*0.01 (0.02)*	*3.51*	*0.00 (0.00)*	*-0.20*	*3.01 (0.00)*	*1.80*	*0.00*
	Settlements	**0.03**		**0.01**		**0.09**		**0.09**		**0.03**		**0.00**		**0.25**	**0.21**	
		0.02 (0.01)	0.76	0.03 (-0.03)	-0.80	0.11 (-0.02)	-0.20	0.02 (0.07)	3.88	0.03 (0.00)	0.00	0.00 (0.00)	1.70	0.21 (0.03)	0.18	0.03
		0.02 (0.00)	*0.21*	*0.06 (-0.05)*	*-0.88*	*0.11 (-0.02)*	*-0.20*	*0.02 (0.07)*	*3.98*	*0.03 (0.00)*	*0.00*	*0.00 (0.00)*	*1.97*	*0.25 (0.00)*	*0.21*	*0.00*
	Water	**0.00**		**0.00**		**0.00**		**0.00**		**0.00**		**0.00**		**0.00**	**0.00**	
		0.00 (0.00)	0.62	0.00 (0.00)	-1.00	0.00 (0.00)	-1.00	0.00 (0.00)	0.00	0.00 (0.00)	0.00	0.00 (0.00)	0.00	0.00 (0.00)	0.00 (0.00)	
		0.00	*0.00*	*0.00 (0.00)*	*0.00*	*0.00 (0.00)*	*0.00*	*0.00 (0.00)*	*0.00*	*0.00 (0.00)*	*0.00*	*0.00 (0.00)*	*0.00*	*0.00 (0.00)*	*0.00 (0.00)*	
	Total 2014	**11.03**		**27.29**		**53.18**		**8.20**		**0.29**		**0.02**		**100.00**	**0.00**	
		11.03 (0.00)	0.00	27.29 (0.00)	0.00	53.18 (0.00)	**0.00**	8.20 (0.00)	0.00	0.29 (0.00)	0.00	0.02 (0.00)	0.00	100.00	49.45	
		12.87 (-1.84)	*- 0.14*	*28.89 (-1.60)*	*-0.06*	*50.81 (2.36)*	0.05	*7.16 (1.03)*	*0.14*	*0.25 (0.04)*	*0.16*	*0.02 (0.25)*	*0.25*	*100.00*	*49.45*	
	Gross Gain	**5.95**		**6.17**		**30.07**	**0.00**	**6.98**		**0.25**		**0.02**		**49.45**	**49.45**	
		5.95 (0.00)	0.00	6.17 (0.00)	0.00	30.07 (0.00)	0.09	6.98 (0.00)	0.00	0.25 (0.00)	0.00	0.02 (0.00)	0.00	49.45		
		7.79 (-1.84)	*-0.24*	*7.77 (-1.60)*	*-0.21*	*27.71 (2.36)*	*0.00*	*5.95 (1.03)*	*0.17*	*0.21 (0.04)*	*0.18*	*0.01 (0.27)*	*0.27*	*49.45*		

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c: Period 4 (1972 - 2014)

		2014													
		For		Woodlands		Savannahs		Croplands		Settlements		Water		Total 1972	Gross loss
1972	Forests	**2.59**		**1.18**		**1.76**		**0.11**		**0.00**		**0.00**		**5.65**	**3.05**
		2.59 (0.00)	0.00	0.99 (0.19)	0.19	2.87 (-1.11)	-0.39	0.46 (-0.35)	-0.77	0.02 (-0.01)	-0.89	0.00 (0.00)	-0.83	6.95 (-1.30)	4.36 (-1.30)
		2.59 (0.00)	*0.00*	*0.94 (0.24)*	*0.26*	*1.82 (-0.06)*	*-0.03*	*0.28 (-0.17)*	*-0.62*	*0.01 (-0.01)*	*-0.82*	*0.00 (0.00)*	*-0.73*	*5.65 (0.00)*	*3.05 (0.00)*
	Woodlands	**5.90**		**20.93**		**33.58**		**3.35**		**0.08**		**0.01**		**63.85**	**42.92**
		5.71 (0.19)	0.03	20.93 (0.00)	0.00	32.46 (1.12)	0.03	5.21 (-1.86)	-0.36	0.18 (-0.10)	-0.54	0.01 (0.00)	-0.22	64.68 (-0.83)	43.75 (-0.83)
		*6.51 (-0.61)**	*-0.09*	*20.93 (0.00)*	*0.00*	*31.38 (2.20)**	*0.07*	*4.84 (-1.49)**	*-0.31*	*0.17 (-0.09)*	*-0.51*	*0.01 (0.00)*	*-0.17*	*63.85 (0.00)*	*42.92 (0.00)*
	Savannahs	**2.52**		**5.16**		**17.73**		**4.67**		**0.19**		**0.01**		**30.28**	**12.55**
		2.71 (-0.18)	-0.07	5.32 (-0.16)	-0.03	17.73 (0.00)	0.00	2.47 (2.20)	0.89	0.09 (0.10)	1.19	0.01 (0.00)	0.50	28.41 (1.88)	10.68 (1.88)
		2.96 (-0.43)	*-0.15*	*7.32 (-2.15)**	*-0.29*	*17.73 (0.00)*	*0.00*	*2.20 (2.47)**	*1.12*	*0.08 (0.11)*	*1.44*	*0.01 (0.00)*	*0.68*	*30.28 (0.00)*	*12.55 (0.00)*
	Croplands	**0.01**		**0.01**		**0.08**		**0.07**		**0.01**		**0.00**		**0.17**	**0.11**
		0.02 (-0.01)	-0.35	0.03 (-0.02)	-0.71	0.09 (-0.01)	-0.13	0.07 (0.00)	0.00	0.00 (0.01)	19.51	0.00 (0.00)	20.78	0.20 (-0.03)	0.14 (-0.03)
		0.01 (0.00)	*-0.21*	*0.03 (-0.02)*	*-0.72*	*0.06 (0.02)*	*0.24*	*0.07 (0.00)*	*0.00*	*0.00 (0.01)*	*29.16*	*0.00 (0.00)*	*31.21*	*0.17 (0.00)*	*0.11 (0.00)*
	Settlements	**0.00**		**0.00**		**0.02**		**0.01**		**0.00**		**0.00**		**0.04**	**0.03**
		0.00 (0.00)	0.26	0.01 (0.00)	-0.76	0.02 (0.00)	-0.19	0.00 (0.01)	2.86	0.00 (0.00)	0.00	0.00 (0.00)	-1.00	0.04 (0.00)	0.04 (0.00)
		0.00 (0.00)	*0.16*	*0.01 (-0.01)*	*-0.83*	*0.02 (0.00)*	*-0.12*	*0.00 (0.01)*	*3.35*	*0.00 (0.00)*	*0.00*	*0.00 (0.00)*	*-1.00*	*0.04 (0.00)*	*0.03 (0.00)*
	Water	**0.00**		**0.00**		**0.01**		**0.00**		**0.00**		**0.00**		**0.01**	**0.01**
		0.00 (0.00)	-0.45	0.00 (0.00)	-0.75	0.01 (0.00)	0.28	0.00 (0.00)	1.87	0.00 (0.00)	0.92	0.00 (0.00)	0.00	0.01 (0.00)	0.01 (0.00)
		0.00 (0.00)	*2.00*	*0.00 (0.00)*	*1.67*	*0.00 (0.01)*	*39.00*	*0.00 (0.00)*	*13.33*	*0.00 (0.00)*	*0.00*	*0.00 (0.00)*	*0.00*	*0.00 (0.01)*	*0.00 (0.01)*
	Total 2014	**11.03**		**27.29**		**53.17**		**8.21**		**0.29**		**0.02**		**100.00**	**58.68**
		11.03 (0.00)	0.00	27.29 (0.00)	0.00	53.17 (0.00)	0.00	8.21 (0.00)	0.00	0.29 (0.00)	0.00	0.02 (0.00)	0.00	100.00	58.96
		12.08 (-1.05)	*-0.09*	*29.22 (-1.94)*	*-0.07*	*51.02 (2.15)*	*0.04*	*7.40 (0.81)*	*0.11*	*0.26 (0.03)*	*0.10*	*0.02 (0.00)*	*0.11*	*99.99 (0.01)*	*58.67*
	Gross Gain	**8.44**		**6.36**		**35.44**		**8.14**		**0.28**		**0.02**		**58.68**	
		8.44 (0.00)	0.00	6.36 (0.00)	0.00	35.44 (0.00)	0.00	8.14 (0.00)	0.00	0.28 (0.00)	0.00	0.02 (0.00)	0.00	58.68	
		9.48 (-1.05)	*-0.11*	*8.29 (-1.94)*	*-0.23*	*33.29 (2.15)*	*0.06*	*7.33 (0.81)*	*0.11*	*0.26 (0.03)*	*0.10*	*0.02 (0.00)*	*0.11*		

Note: Each cell is subdivided into three rows and two columns of numbers. Left column of each cell: the first row contains bolded numbers that represent the actual (observed) proportions of inter-categorical transitions (persistence and transitions) of the landscape. The second row represents the expected percentage of land under random processes of gain (named Expected (+)) calculated using Equation 2, where figures in round parentheses are equal to the observed proportion minus the one expected (named Difference (+)). The third row contains italicized numbers representing the expected proportion of land under random processes of loss (named Expected (-)) calculated using Equation 3, where numbers within round parentheses represent the observed proportion minus the expected one (named Difference (-)). Extreme right column of the table contains the Loss-to-persistence ratio ($L_{(-)}$) while the extreme row is the Gain-to-persistence ratio ($G_{(+)}$). Numbers highlighted in gray represent systematic gain transitions; starred numbers are the systematic loss transitions.

5.4. Discussion

5.4.1. Historical trends and processes of LUCC in the Mo basin

The LUC mapping showed that natural vegetation dominated the Mo basin for all the observed periods (1972, 1987 and 2000 and 2014). At period (1972 and 1987), most parts of the Mo landscapes were greener and dominated by woodlands and savannahs. Agricultural patches (360 ha over about 149,000 ha of the whole basin) were merely located around scattered human settlements in the landscapes, especially along the main roads. Despite this dominance of natural vegetation over time, woodlands showed acute areal loss while forests and savannahs substantially increased along the study period. Similar decrease in natural vegetation was observed in the adjacent landscapes of the Kara River basin (Badjana et al., 2014) and the Northern Togo (Folega et al., 2015; Folega et al., 2014b). Agricultural expansion associated with fuelwood collection were reported as the prominent factors of natural vegetation loss. Although it is of small-scale, agriculture in the study area was regarded as the main reason of deforestation and forest degradation. This result aligned with the findings of Lindstrom et al. (2012) who reported the role of the small-scale farming in forest cover loss. In addition, as reported by Sassen et al. (2015), firewood collection plays a significant role in DFD, even in PA. Regarding the water bodies, the significant variations observed during the two last periods could be explained by the open water availability in the rivers and dams influenced by either the siltation of Aleheride reservoir constructed since 1970s and the construction of a second reservoir in 2004 (DRHE, 2004), or the inter-annual rainfall variability.

Although the Mo basin still exhibits high potential of wild landscapes attributable to its lower population compared to other regions of the country (DGSCN, 2010), its high proportion of PA and the landscape ruggedness, human concentration in UPA induced much more degradation and fragmentation in those landscapes. In addition to the quantitative LUCC processes (i.e. amount or net change), the qualitative (i.e. location or swapping) changes were observed. This suggested that LUC types experienced much more spatial transformation than indicated by the net change detection. Sole the forest category exhibited quite stable values of swap changes when the extent of analyses was large (1972 - 2014). For this overall period, the fair swaps for savannahs and woodlands suggested that location changes occurred lesser than as revealed by the transition periods. The extent of transitional period (number of years) is thus an important factor influencing the detection of swapping processes; the longer the period the lower the likelihood to observe LUCC occurring in between the observation dates. This suggests the importance of high temporal resolution (regular and shorter observational periods) for sharper monitoring of the landscape dynamics. In the current study, the mapping approach exclusively relied on the landscape conditions at the exact passing times of the satellite. Such analysis approach could introduce biases in the assessment of the real LUC dynamics since it had limitations in verifying the potential changes that occurred in between two successive dates (Braimoh & Vlek, 2005; Garedew, 2010). Nevertheless, the approach was helpful in analysing the historical land dynamics in a data scarce situation such as that of the Mo basin.

In general, the reliability of these observed statistics and trends was satisfactory with regard to the assessment accuracies (69 - 92 %) and Kappa indices (0.6 - 0.9). As pointed out by several authors (Aguirre-Gutiérrez et al., 2012; Leh et al., 2013; Monserud & Leemans, 1992), these accuracy levels are satisfactory for monitoring and modelling of LUCC processes. In addition, the overall accuracies of the transition maps were of 62.9, 82.7, 83.3 and 63.3 %, respectively for the 3 and overall transitional periods. These derived transition maps from the individual LUC maps were therefore considered reliable (Were et al., 2013). Further, the producer and user accuracies not only ascertained the above-mentioned accuracies but the reported classification errors that could be introduced by the spectral confusion between LUC types, as indicated by (Were et al., 2013). In addition, as it is the case in the Mo basin, Lu (2006) mentioned that other factors such as complex topographical and vegetation patterns could induce complexity in spectral responses rendering cumbersome the classification processes. Furthermore, misclassification errors could be introduced via data used for validation since the resolutions of the historical reference data for the past LUC maps were relatively of poor resolution for small-scale breakdown and could introduce biases (Verburg et al., 2013). However, they were of great interest for such analyses in the data-scarce area of the Mo basin. In this regard, Houet et al. (2010) pointed that the ongoing challenge related to data availability compels to the use of multisource data for assessing landscape dynamics.

5.4.2. Landscape dynamics, land legislation and implications

Regardless of the land protection status and the transition periods, the processes of change affected approximately 50 % of the lands in the whole basin between 1972 and 2014. Persistent natural vegetation and processes of vegetation growth are becoming scarce in both PA and UPA. With a focus on the effectiveness of law enforcement on biodiversity conservation, protection law played an important role for land conservation, though vegetation loss occurred in the wildlife reserves and parks. In fact, illegal incursions were noticed in PA, especially the Wildlife Reserve of Aledjo (Wala et al., 2012) and the national park of Fazao-Malfakassa, within the neighbourhoods of Alombe and along Mo and Bouzalo rivers since 1978 (Aboudou, 2012). Despite the conservation measures, other

PA of Northern Togo experienced similar loss and degradation with more acute levels (Dimobe et al., 2014; Folega et al., 2010; Folega et al., 2012). The less degradation situation of the PA in the Mo basin is attributable to the private status devolved to FMN Park especially during the period 1990 - 2015. Though it occurs at low rate compared to the situation in UPA, the natural vegetation loss should be of major concern to counteract the degradation trend. As such, efforts are explicit towards the law reinforcement for PA, especially the case of FMN Park, which gained much attention through two major concrete actions: (i) the assignment of the park to a private monitor and manager for 25 years (1990 - 2015) and (ii) redefinition of new boundaries and enhancement of its conservation role.

These efforts should focus on addressing the challenges of LD in the area through three major mechanisms. First, there should be an effective law enforcement and promotion of sustainable resource in both PA and UPA. In accord with Paudel et al. (2015), institutional operationalisation, especially at local level, regarding the land issues is needed to ensure a holistic framework for sustainable land use and conservation. Next, a clear definition of the role of all involved stakeholders from the formal to the informal as well as the public-private partnership in rural communities should support this institutional setting (Nolte et al., 2013). This could ensure a careful consideration of the real needs of all stakeholders in order to avoid breaking the law, as reported in several studies (Tumusiime et al., 2011; Vedeld et al., 2012). Meanwhile, common lands outside the PA might gain much more attention towards a sustainable use in order to avoid issues related to common resources. Several studies reported the strategies of payments for environmental services (PES) and the REDD+ as incentives to abate LD in rural areas (Mattsson et al., 2012; Nakakaawa et al., 2010). Lastly, the reduction of the dependence of rural households on forest products could be of positive effect in the process of landscape conservation in the Mo basin. Strategies for improving their socio-economic conditions could therefore include diversification of income-generated activities such as the promotion of orchards and cashew plantations, energy woodlots and agroforestry systems in order to support socio-ecological development (Mbow et al., 2014a; Mbow et al., 2014b).

5.5. Conclusion

The reliance on proxies such as Landsat archives was useful to monitor landscape transformation and its factors. In this study, these historical Landsat data provided an assessment of the process of vegetation degradation in rural areas of the Mo river basin. The study provided more light on the extent, location and rate of rural land transformation, mainly deforestation and forest degradation (DFD). The results revealed that the observed trend in the study area are the intensive decline of woodlands associated with the increase in savannahs, forests and cultivated areas. Natural vegetation showed a decreasing trend from 99 % in 1972 to 91 % in 2014. This significant change in the natural land cover quality and extent were due to agricultural expansion and wood extraction in both protected and non-protected areas. The trend in the landscape dynamics indicated a "savanisation" process though there was an improvement of forest cover. The net balance of natural vegetation changes indicated an overall loss mostly due to woodland decline. However, the Mo basin still provides wide opportunities for forest protection and natural landscape integrity. This information may be of practical relevance in guiding managers and policy makers for reversing the loss of natural vegetation through the formulation and implementation of new strategies for integrated land management. Restoring the degraded areas and increasing the awareness are essential for maintaining viable landscapes and biological conservation, supporting sustainable livelihood, and mitigating climate change. This study showed that timely and urgently needed spatial information could be drawn from satellite images at landscape level to detect hotspot areas where efforts should concentrate for conservation and restoration processes. With regard to the ecological and economic importance of the Mo river basin and the surrounding lands, further analyses of the major LUCC trajectories, their underlying factors, and their potential impacts on the ESS could offer further understanding of the landscape dynamics towards alternative pathways of sustainable development of the basin.

CHAPTER 6: IMPACTS OF LUCC ON ECOSYSTEM SERVICES AT LANDSCAPE AND SITE-SPECIFIC LEVELS IN THE MO RIVER BASIN [4]

6.1. Introduction

During the recent decades, there is a growing concern in scientific communities about how much land has changed and how this change is going to affect the future of land resources, and the provision of ecosystem services (ESS). Lands provide multiple ESS, including medium for living organisms, soil fertility for agricultural production, regulation of the hydrologic cycle, and mitigation of climate change through carbon sequestration (MEA, 2005; Oladele & Braimoh, 2011; Wiesmeier et al., 2013b; Winowiecki et al., 2015). However, the human impacts on terrestrial ecosystems have induced landscape fragmentation and loss of land capability to provide services and support livelihood (Balthazar et al., 2015; Schleuning et al., 2011). Regardless of the scale, the landscape dynamics occur continuously, making it difficult to monitor their extent and impacts, even in protected areas that have a devoted role of conservation (Castro et al., 2015; Damnyag et al., 2013; Folega et al., 2014b; Traoré et al., 2012; Vedeld et al., 2012).

Under climate change conditions, land management and use as adaptation and mitigation options could play a major role in ESS provision, especially the magnitude of sediment loss, SOC and TN storages as well as other landscape services (Guillaume et al., 2015; Lacoste et al., 2015). For instance, it has been shown that land conversion from natural vegetation into croplands induced landscape degradation and subsequent decline

[4] This chapter is an ongoing manuscript to be submitted.

in soil nutrients including SOC and TN (Biro et al., 2013; Braimoh & Vlek, 2004a; Were et al., 2015). On the other hand, sediment loss is shown to be significantly responsive to landscape patterns affected by LUCC (Zhou et al., 2014) as well as the management conditions (Ouyang et al., 2010; Qiao et al., 2015). This suggests that LUCC could have limited impacts on landscape configuration and functions unless good management practices are developed (Labriere et al., 2015; Zhou et al., 2014a). Yet, it is unclear what exactly controls nutrient stocks and balance in multifunctional landscapes under constant dynamics. There is little certainty regarding soil conditions under different LUCC trajectories in promoting sustainable directions to land management.

Recently, earth observation data have brought new insights into the approaches for understanding the LUCC, monitoring and assessing LD and its impacts on ESS at various spatio-temporal scales. In addition, by combining contemporary field observations with legacy information, understanding of potential impacts of landscape dynamics on ecosystem structure and services have been significantly improved, especially in data-scarce regions. Furthermore, landscape fragmentation analyses have emerged from the use of these tools and data with landscape metrics as ecological indicators to quantify the composition and spatial configuration of landscapes under constant change (Mander & Uuemaa, 2010; Peng et al., 2010; Uuemaa et al., 2013). Thus, as an essential approach in quantifying landscape patterns, landscape metrics could help in the analysis of changes in LUC patterns and related ecological effects (Uuemaa et al., 2013; Walz, 2011).

Although some recent studies have highlighted LD in Togo through the analyses of DFD and human disturbances (Badjana et al., 2015; Folega et al., 2015; Folega et al., 2014b; Wala et al., 2012) and of cultivation effects on soil nutrient availability at farm plot levels (Kintche et al, 2013; Sebastia et al, 2008), the impacts of such changes on the spatial patterns of soil nutrients and erosion at the landscape level have not received much research attention. In the current study, such data are produced through a comprehensive analysis of the potential impacts of LUCC in relation to *in situ* environmental variables, on the ESS at landscape level. Based on the concept of landscape ecology which studies the relationship between spatial patterns and ecological processes at landscape level, this study argues that changes in landscape conditions affect the ESS and the involved ecological processes. Therefore, data from post-classification of satellite legacy information combined with the analysis of landscape metrics and field surveys were used to assess the potential impacts of LUCC trajectories on ESS (landscape configuration, soil erosion, SOC and TN) in the Mo basin. As the study emphasises on the interdisciplinary approach to spatially explicit the landscape dynamics and impacts on ESS, its ultimate goal is to provide useful information for adapted management and conservation of land resources in the Mo basin and other landscapes.

6.2. Materials and Methods

6.2.1. Study area: land management challenges and land degradation

Land resources in Togo fall under two major categories based on protection status: PA including national parks, wildlife reserves and forest reserves; and UPA generally including common lands with community protection systems. Challenges of sustainable landscapes often emerge from the misuse and mismanagement that led to the degradation, deforestation and resources scarcity in the Central Togo, especially in some parts of the Mo basin. Resource collection and land allocation decision-making are often based on the resource availability, proximity and accessibility rather than planning schemes. The disregard of legislation on PA, results in illegal incursions for resource collection leading to the fragmentation and degradation of their ecosystems, as it is the case in the three PA of the basin. Important land transformations are still occurring in the basin mainly due to small-scale farming and forest-resource collection (Fontodji et al., 2011; Kokou et al., 2009; Wala et al., 2012) resulting in an increase of LD (Figure 6.1).

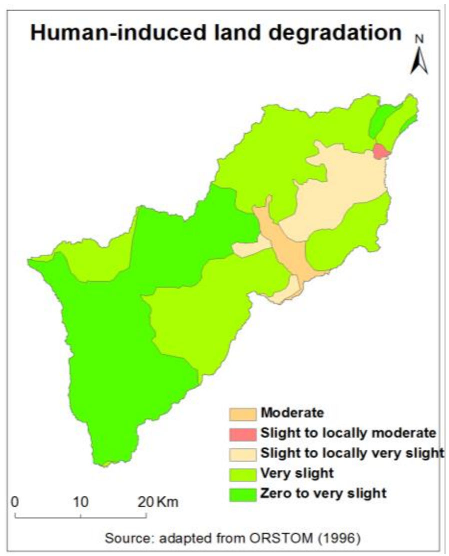


Figure 6.1. Human-induced land degradation in the Mo River basin

The prominent environmental issues are LD due to overgrazing, unsustainable agricultural land use, fuel wood harvesting and charcoal production (Dourma et al., 2009; Wala et al., 2012; Woegan, 2007). Illicit incursions for hunting and tree logging in PA are also concerns that cause conflicts between land users and state agencies protecting lands (Aboudou, 2012). Soil fertility loss and erosion have been mentioned among these environmental challenges (MERF-Togo, 2010). Foremost of the land uses in the area is small-scale subsistence farming, pasture lands, protected areas and built-up areas. Agriculture is the most important sector of the economy, employing 70 % of active population and accounts for over 40 % of GDP (ADB & ADF, 2011).

6.2.2. Analysing the LUCC trajectories

LUC maps for 1972, 1987, 2000 and 2014 were obtained from the classification of historical Landsat images covering the Mo basin (Chapter 5 Section 5.2). The transition maps between two periods were obtained from the pairwise cross tabulation from the four LUC maps. Thus, for each resulting map, the transition/conversions between different LUC types were reclassified into maps of six categories with less attention to water and settlements classes. LUC transition category between two dates refers to a succession of LUC types for a given landscape unit over more than two observational dates (Braimoh & Vlek, 2004b; Zhou et al., 2008). Following Rojas et al. (2013) and adapting to the definition of deforestation and forest degradation (DFD) from Kyoto protocol (GOFC-GOLD, 2009), land cover transitions were clustered into transition categories based solely on the processes involved.

Transitions among natural vegetation types (i.e. savannah, woodland and forest) and between natural vegetation and human-induced landscapes (e.g. natural vegetation and cropland) (Braimoh & Vlek, 2004b) were analysed. Any change into more (or less) vegetation cover was categorised as vegetation improvement (or decline). For instance, the conversion of an initial forest into a woodland or savannah is considered a loss of vegetation quality. Conversely, a situation is categorised into vegetation gain if the initial pixel was of less cover than its final cover. The conversion of savannah into a woodland or forest pixel is a typical illustration of the vegetation gain. Similarly, the transition between natural vegetation and cropland were analysed. For instance, any transition between natural vegetation and croplands was qualified as agricultural deforestation (or land abandonment) if the initial state was a typical natural vegetation converted into croplands (or inversely). Finally, the proportions of the transition coverages were computed as the proportions to the total area of the basin. The influence of PA was analysed by integrating the layer of the PA network extracted from the topographic map of Togo (IGN, 1986) updated from the last redefinition of national PA network. These latter transition proportions in either UPA or PA were reported as relative proportions to the areas of the UPA and PA, respectively.

6.2.3. Analyses of the impacts of LUCC trajectories on ESS

a) Analysis of landscape fragmentation using landscape metrics

Patterns in wild and human-impacted systems are often characterized by not only long-term spatial changes in vegetation cover but also landscape metrics (LMs). These metrics are used for the identification and measurement of the landscape dynamics in terms of habitat quality assessment in identifying and measuring landscape dynamic (Kang et al., 2013; Mander & Uuemaa, 2010; Renetzeder et al., 2010; Schindler et al., 2013; Uuemaa et al., 2013). Common usage of the term “landscape metrics” refers exclusively to indices developed for categorical map patterns. Numerous metrics have been developed to quantify such categorical map patterns, but they fall into two general categories: those quantifying the composition of a map without reference to spatial attributes, and those that quantify the spatial configuration of a map, requiring spatial information for their calculation (McGarigal et al., 2002; Smith et al., 2009). The latter category is of interest in this study. As selection criterion suggested by Schindler et al. (2013), previously published LM in similar studies were selected in order to avoid redundancy related to LM plethora. In addition, many metrics are usually selected to avoid the assessment of landscape fragmentation based on a single measure rather than on the aggregation from multiple LM. In this study, five metrics were therefore selected (Table 6.1) (Wang et al., 2013a; Wu et al., 2014; Zhang et al., 2013) to evaluate landscape fragmentation at class and landscape levels using FRAGSTATS 4.2.1 (McGarigal & Ene, 2013). Using 8 cell neighbourhood rule without any sampling method, LMs were calculated for the three LUC types (viz. forest, woodland and savannah) for the four observation periods.

Table 6.1. Land cover-based landscape metrics for approximating ESS change (McGarigal et al., 2002; McGarigal & Marks, 1995)

Indices	**Meaning of index -Measured ESS**
Patches number (NP)	NP ≥ 1, without limit. NP = 1 when the landscape contains only 1 patch of the corresponding patch type. Measure of the extent of class fragmentation. *Under landscape fragmentation, NP was hypothesised to increase with time.*
Patch density (PD)	PD expresses number of patches on a per unit area basis that facilitates comparison among landscapes of varying size. PD > 0. *Under landscape fragmentation, PD was hypothesised to increase with time.*
Largest patch index (LPI)	Largest patch index quantifies the percentage of total landscape area comprised by the largest patch. As such, it is a simple measure of dominance. 0 < LPI ≦ 100; LPI approaches 0 when the largest patch in the landscape is increasingly small. *Under landscape fragmentation, LPI was hypothesised to decrease with time.*
Patch cohesion index – COHESION	Patch cohesion index measures the physical connectedness of the corresponding patch type. 0 < COHESION < 100; COHESION approaches 0 as the proportion of the landscape comprised of the focal class decreases and becomes increasingly subdivided and less physically connected. *It was hypothesized that a single LUC patch would split into smaller and sparse patches creating lower COHESION towards 0.*
Aggregation index- AI	0 ≦ AI ≦ 100. Aggregation index is calculated from an adjacency matrix, which shows the frequency with which different pairs of patch types (including like adjacencies between the same patch types) appear side-by-side on the map. *Under landscape fragmentation, it was hypothesized that a single LUC patch would split into smaller and adjacent patches creating higher AI towards 100*

b) Analysing the effects of LUCC trajectories on SOC and TN

The effects of LUC conversion on SOC and TN were analysed through the comparison of the average values of these properties under different LUCC trajectories. Data on SOC and TN at two soil depths (0 – 10 and 10 – 30 cm), were obtained from soil samples collected during the field surveys (See Chapter 3 Section 3.2 and Chapter 4 Section 4.2). The geographical coordinates of the sample sites were overlaid on the LUC maps of 1987, 2000 and 2014 in order to define the LUCC trajectories at sample sites. The LUCC trajectories were defined by successive transitions between the LUC types between 1987 and 2014, assuming that no different change occurred in between two successive observational dates (Braimoh & Vlek, 2004a; Braimoh & Vlek, 2005). Seven major LUCC trajectories were observed at the sample sites (Table 6.2). Three performance types were defined to analyse the effects of land performance on soil conditions (Table 6.2). Accordingly, any trajectory inducing an improvement in LUC cover was categorised as "improvement" whereas "decline" indicated that the land performance is negative based on the trajectories. The unchanged natural vegetation (persistent forests, woodlands and savannahs) were named "unchanged".

Table 6.2. LUCC trajectories at sample sites according to Braimoh and Vlek (2005)

	Land cover in observation dates			LUCC trajectories	Performance types
N	1987	2000	2014		
1	Croplands	Natural vegetation	Natural vegetation	Abandonment	Improvement
2	Natural vegetation	Natural vegetation	Natural vegetation	Non-cultivated	Unchanged
3	Savannahs	Forests	Forests	Old regrowth	Improvement
4	Natural vegetation	Natural vegetation	Croplands	Recent croplands	Decline
5	Forests	Forests	Savannahs	Recent degradation	Decline
6	Savannahs	Savannahs	Forests	Recent regrowth	Improvement
7	Forests	Savannahs	Savannahs	Old degradation	Decline

c) Effects of LUCC trajectories on soil loss potential

The effects of LUCC trajectories on soil loss were estimated at the soil sample sites by overlaying their geographical coordinates on the soil loss map of 2014. Based on the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997), a GIS-based soil loss map was developed for the study area. Equation 6.1 is the RUSLE expressing the annual soil loss amount (Mg $ha^{-1}y^{-1}$) for a specific location:

$$A = R \times K \times LS \times C \times P \qquad \text{(Equation 6.1)}$$

The RUSLE factors were produced at 30 m resolution to match Landsat data. The processes for the development of spatial layers for K, R, C and LS factors are detailed in Chapter 3 Section 3.2 and Chapter 7 Section 7.2.2**.** P is the management (such terracing, stone lines, etc.) support practice factor. As in several studies, P factor was set to 1 for the whole study area, since there were no noticeable supporting practices.

6.2.4. Other datasets and data analyses

Terrain data from the SRTM - DEM were used to derive topography-based indices that could be potential environmental factors affecting landscape conditions. The processing and the hypothesised effects of these factors are described in Chapters 3 and 4. In addition, proximity variables (distance to roads and to villages) were derived from the last demographic census database (DGSCN, 2010) using Euclidean Distance analyses in GIS. The potential effect of land protection regime on ESS provision and land conservation was assessed by integrating a layer of PA network of the Mo basin in the analytical framework.

Average values for the environmental variables were analysed using descriptive statistics (mean and standard deviation). ANOVA was performed to compare the patterns of these soil parameters according to LUCC trajectories at site level. Further, pairwise correlation at 95 % confidence interval was used to examine the potential interactions between SOC and TN contents, environmental variables, and soil loss amount at the different sites. Exploratory data analyses were performed using principal component analyses to further examine the relationships between soil conditions and other environmental variables under the different LUC trajectories in the Mo basin.

6.3. Results

6.3.1. Patterns of the major land cover transitions

The statistics on LUC types and LUCC are provided in Table 5.2a, b, c and d and Figures 5.1 and 5.2 (Chapter 5). The clustering of these LUCC into conversion categories and trajectories showed that non-cultivated lands and vegetation growth areas were widely dominant in the landscapes during all the periods. These categories declined progressively southward with the occurrence of human-transformed landscapes. This shaped the prevalence of natural vegetation and forest growth in PA whereas degradation and cropland expansion processes dominated UPA (Figure 6.2 and Figure 6.3). For instance, forest growth declined from 16 % to 5 % between the first and the third periods, respectively whereas the conversion into agricultural lands were more acute in UPA, increasing from 2 % for the period 1972-1987 to 14 % for the period 2000-2014. Meanwhile, permanent agricultural lands were more observed outside the UPA.

All these land transformation processes occurred at different magnitudes irrespective of the land protection regime, as indicated in Table 6.3. The annual rates of natural vegetation decline in PA were higher during the first and third periods (1.2 % and 2.1 %, respectively). Meanwhile, the highest rate was in UPA during the second period (1.6 %). The process of vegetation regrowth from agricultural land abandonment is far low to compensate this trend of agriculture-induced cover loss. During the overall period, the rate of land conversion processes within PA and UPA were higher for the degradation of natural vegetation (0.9 %) and agricultural deforestation (0.3 %). The implication is that a greater part of the landscape experienced negative transformations due to an increasing level of human appropriation of the natural landscapes.

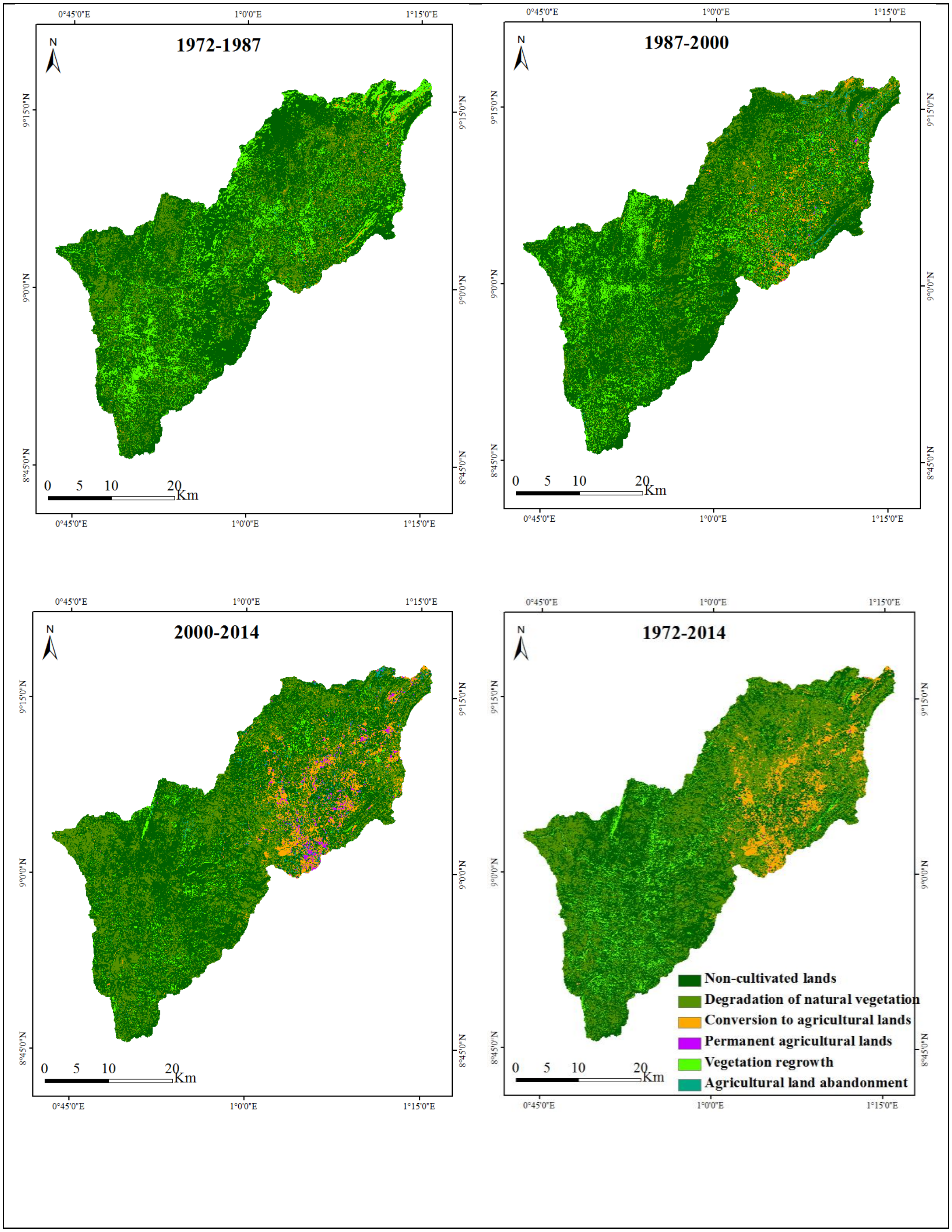


Figure 6.2. Land conversion categories for the four study periods

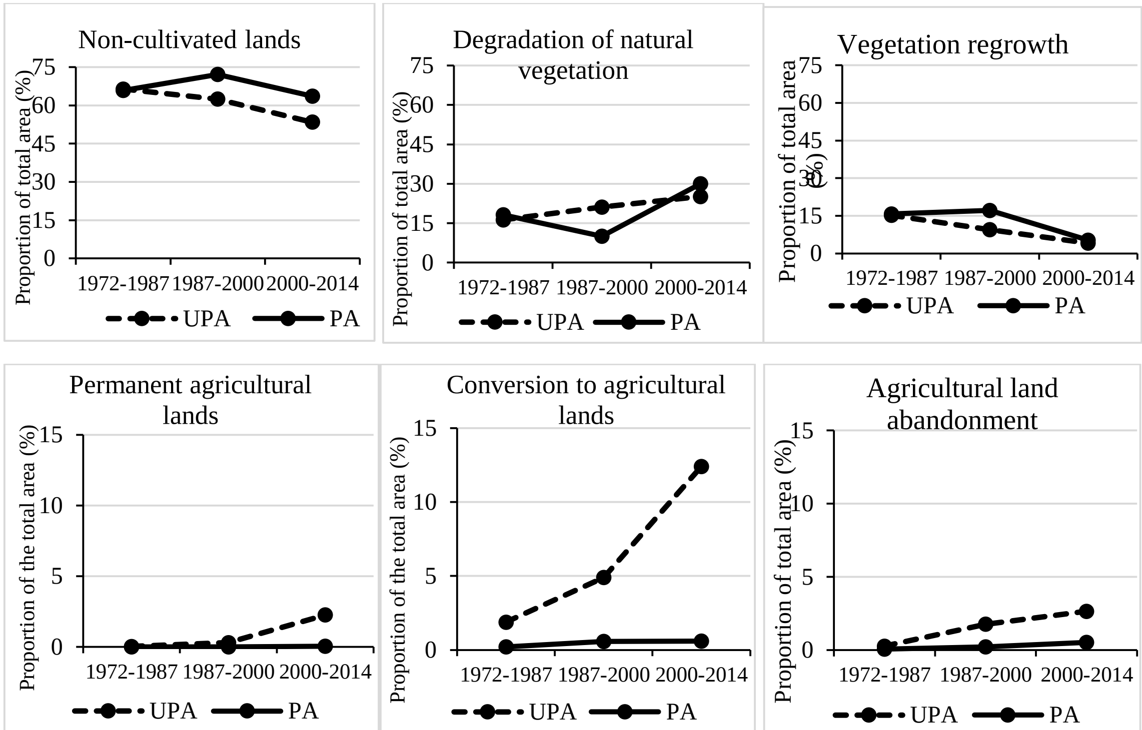


Figure 6.3. Land trajectory categories according to land protection

Table 6.3. Rates of conversion processes according to land management regime

Mean annual rate (%)	1972-1987		1987-2000		2000-2014		1972-2014	
	PA	UPA	PA	UPA	PA	UPA	PA	UPA
Degradation of natural vegetation	1.215	1.083	0.770	1.627	2.142	1.797	0.731	0.924
Conversion to agricultural lands	0.014	0.125	0.045	0.377	0.043	0.886	0.015	0.342
Vegetation regrowth	1.047	1.016	1.316	0.728	0.376	0.295	0.244	0.101
Agricultural land abandonment	0.005	0.018	0.017	0.136	0.038	0.189	0.001	0.003

6.3.2. Transitions among natural vegetation categories

Table 6.4a shows that the transformation among natural vegetation induced significant quality decline. For both PA and UPA, about 45 %, 41 % and 61 % of the total basin experienced quality decline during the 3 periods, respectively. The overall change proportion showed that UPA underwent more vegetation decline than PA did. On the other hand, regardless the land protection status, the conversion to more vegetation cover occurred at highest proportion during 1987-2000 (about 41 %) whereas the proportions were 40 %, and 19 % for the first and third periods, respectively. For the overall period, the proportions of transition to less vegetation (40 % and 33 % for UPA and PA, respectively) were far higher than the ones of transition to more vegetation (10 % and 17 % for UPA and PA, respectively). This suggests that the Mo River basin experienced a decrease in vegetation cover quality despite its important network of protected areas.

Table 6.4a. Detailed transitions among natural vegetation types

	Transition inducing vegetation cover decline			Transition inducing vegetation improvement		
		UPA	PA		UPA	PA
1972-1987	Forest to woodlands	*3.979*	*2.321*	Woodlands to forests	4.195	2.758
	Forest to savannahs	0.447	0.266	Savannahs to woodlands	15.767	15.797
	Woodlands to savannahs	*17.190*	*20.938*	Savannahs to forests	0.662	0.502
	Total	**21.616**	**23.525**	**Total**	**20.624**	**19.057**
1987-2000	Forest to woodlands	3.419	1.435	Woodlands to forests	3.945	5.755
	Forest to savannahs	0.543	0.273	Savannahs to woodlands	10.805	19.846
	Woodlands to savannahs	22.128	11.425	Savannahs to forests	0.211	0.894
	Total	**26.089**	**13.134**	**Total**	**14.960**	**26.494**
2000-2014	Forest to woodlands	1.628	2.534	Woodlands to forests	4.520	3.647
	Forest to savannahs	1.207	1.504	Savannahs to woodlands	3.611	4.489
	Woodlands to savannahs	24.806	29.691	Savannahs to forests	1.422	2.143
	Total	**27.641**	**33.729**	**Total**	**9.553**	**10.279**
1972-2014	Forest to woodlands	1.403	0.915	Woodlands to forests	5.719	6.116
	Forest to savannahs	2.340	1.074	Savannahs to woodlands	2.896	7.878
	Woodlands to savannahs	36.088	30.580	Savannahs to forests	1.865	3.315
	Total	**39.831**	**32.570**	**Total**	**10.480**	**17.308**

Note: Indicated values are transition percentages of the total areas of PA and UPA

6.3.3. Transitions between natural vegetation and cultivated lands

Croplands replaced natural vegetation by about 3 %, 5 % and 13 %, for the 3 periods, respectively (Table 6.4b). These increasing proportions indicate a continuous agricultural land expansion, especially during the third period. Most of the transitions to croplands occurred in UPA although PA is also crop-affected (lower than 1 % for all periods). Transitions to croplands were higher than the reverse process (croplands to natural vegetation with 0.3%, 2 % and 3 %, respectively for the 3 periods). For the overall period 1972-2014, cumulative proportions of the conversion to less vegetation (72 %, Table 6.4a) and to croplands (15 %, Table 6.4b) indicated that the whole basin experienced significant changes towards a decline of vegetation cover (about 87 % over 1972-2014). The counter-balance proportion of lands experiencing improvement from natural vegetation (28 %, Table 6.4a) and agricultural land abandonment (0.2 %) were quite low and corroborated the general trend of vegetation decline in the study area.

Table 6.4b. Transitions between natural vegetation and agricultural lands.

	Transition natural vegetation to croplands			Transition croplands to natural vegetation		
		UPA	PA		UPA	PA
1972-1987	Forests to croplands	0.016	0.004	Croplands to forests	0.001	0.001
	Woodlands to croplands	0.839	0.095	Croplands to woodlands	0.096	0.039
	Savannahs to croplands	1.394	0.183	Croplands to savannahs	0.145	0.020
	Total	**2.249**	**0.282**	**Total**	**0.242**	**0.060**
1987-2000	Forests to croplands	0.016	0.005	Croplands to forests	0.015	0.007
	Woodlands to croplands	1.821	0.212	Croplands to woodlands	0.529	0.097
	Savannahs to croplands	2.958	0.284	Croplands to savannahs	1.425	0.172
	Total	**4.795**	**0.501**	**Total**	**1.969**	**0.276**
2000-2014	Forests to croplands	0.079	0.037	Croplands to forests	0.043	0.064
	Woodlands to croplands	3.960	0.232	Croplands to woodlands	0.164	0.053
	Savannahs to croplands	8.181	0.265	Croplands to savannahs	2.652	0.361
	Total	**12.220**	**0.534**	**Total**	**2.859**	**0.478**
1972-2014	Forests to croplands	0.175	0.026	Croplands to forests	0.006	0.015
	Woodlands to croplands	5.938	0.249	Croplands to woodlands	0.009	0.008
	Savannahs to croplands	8.327	0.288	Croplands to savannahs	0.122	0.024
	Total	**14.440**	**0.563**	**Total**	**0.137**	**0.047**

Note: Indicated values are transition percentages of the total areas of PA and UPA

6.3.4. Stratification of the LUCC trajectories according to biophysical factors

Table 6.5a indicated that at lower slopes, natural vegetation decreased from 55 % during the first to 47 % during the third period. The worst degradation of natural vegetation in low slopes (24 %) occurred during 2000-2014. Meanwhile, for all slope classes, the natural vegetation growth exhibited a marked decreasing trend during all transition periods. From less than 1 to 7 %, agricultural encroachment exponentially increased for the lands with slopes < 15 % while steeper slopes showed erratic agricultural transformation. Though permanent croplands showed decreasing proportions with increasing slopes, an increase was revealed in their proportions for all slope classes over time. Agricultural land abandonment increased during the two recent periods with a greater preference for slopes < 15 %. In broad, all the conversion processes followed a decreasing slope gradient i.e. land transformation preferentially occurred in the lower slopes.

On the other hand, proximity analysis of the spatial configuration of LUCC trajectories (Table 6.5b) showed that the non-cultivated lands as well as vegetation growth areas decreased over time for all distance-to-road classes. For instance, in the first 1000 m along roads, natural vegetation (from 13 % to 11 %) and the vegetation regrowth (from 4 % to less than 1 %) decreased during the first and the third periods, respectively. In contrast, permanent croplands showed an increasing preference for lands near to the main roads (< 3 km) whereas agricultural abandonment were most prevalent in the second and third buffers. In general, human-affected lands mostly occurred close to roads (< 3 km) while natural lands (vegetation growth and non-cultivated lands) are more prominent in farther distances (> 3 km). This suggests that the greater the distance from the main roads, the better the performance of the natural processes of land transition.

Similarly, most of land transition processes occurred within distances up to 5 km (Table 6.5c). In relation with the trends observed for the road proximity analyses, the proportions of all transition processes were high for natural processes in the distant lands. Inversely, as distance from villages increased, human disturbances decreased. For instance, vegetation growth in the first buffer was 4 %, 9 and 4 % for the 3 classes of distance, respectively. In contrast, in the first 1 km zone, conversion to agricultural lands increased from 1 % during 1972-1987 to 3 % and 6 % for the second and third transition periods, respectively, indicating that agricultural lands increased more consistently in the areas adjacent to villages.

Table 6.5a. Land transition processes according to slope classes

Transition periods	1972-1987			1987-2000			2000-2014			1972-2014		
Slope classes	< 15 %	15- 25 %	> 25 %	<15 %	15-25%	> 25 %	<15 %	15-25 %	> 25 %	< 15 %	15-25 %	> 25 %
Non-cultivated lands	54.90	10.24	0.94	55.12	10.59	1.11	47.24	9.69	1.10	40.26	8.47	0.87
Degradation of natural vegetation	15.41	1.68	0.05	13.33	2.53	0.21	23.51	3.72	0.12	30.32	4.60	0.18
Conversion to agricultural lands	0.78	0.28	0.06	2.71	0.21	0.01	6.69	0.32	0.03	7.74	0.34	0.03
Permanent agricultural lands	0.02	0.00	0.00	0.14	0.02	0.00	1.21	0.03	0.01	0.07	0.00	0.00
Vegetation growth	12.33	2.73	0.39	11.59	1.32	0.04	3.48	1.00	0.16	5.10	1.53	0.36
Agricultural land abandonment	0.17	0.01	0.00	0.73	0.28	0.06	1.49	0.18	0.01	0.10	0.01	0.00

Table 6.5b. Land transition processes according to distance to main roads

Transition periods	1972-1987			1987-2000			2000-2014			1972-2014		
Distance to roads	0-1 km	1-3 km	3-5 km	0-1 km	1-3 km	3-5 km	0-1 km	1-3 km	3-5 km	0-1 km	1-3 km	3-5 km
Non-cultivated lands	13.37	25.18	21.45	12.70	23.84	19.84	10.61	21.02	16.78	9.12	17.02	11.80
Degradation of natural vegetation	3.75	7.86	6.09	4.40	10.50	8.45	4.11	10.03	10.02	6.16	15.78	15.83
Conversion to agricultural lands	0.84	1.34	0.60	2.11	3.30	1.71	4.91	7.62	4.70	6.03	9.04	5.18
Permanent agricultural lands	0.03	0.01	0.00	0.21	0.20	0.06	1.24	1.56	0.60	0.12	0.06	0.04
Vegetation growth	4.33	8.95	5.75	2.36	4.37	3.33	0.72	1.48	0.80	1.00	1.50	1.09
Agricultural land abandonment	0.21	0.15	0.07	0.74	1.27	0.60	0.94	1.76	1.08	0.11	0.08	0.04

Table 6.5c. Land transition processes according to the distance to villages

Transition periods	1972-1987			1987-2000			2000-2014			1972-2014		
Distance to villages	0-1 km	1-3 km	3-5 km	0-1 km	1-3 km	3-5 km	0-1 km	1-3 km	3-5 km	0-1 km	1-3 km	3-5 km
Non-cultivated lands	13.71	28.38	18.33	12.83	27.51	17.24	9.59	23.17	14.78	8.06	17.96	11.11
Degradation of natural vegetation	4.43	9.61	5.13	4.20	11.20	6.42	4.81	12.59	8.22	7.10	19.37	12.72
Conversion to agricultural lands	0.93	1.27	0.31	2.51	3.14	1.08	5.79	8.35	2.77	7.28	9.42	3.09
Permanent agricultural lands	0.04	0.01	0.00	0.25	0.13	0.05	1.60	1.24	0.36	0.11	0.06	0.02
Vegetation growth	3.99	9.29	4.18	2.71	5.44	2.94	0.49	1.49	1.17	0.64	1.80	1.04
Agricultural land abandonment	0.20	0.14	0.04	0.80	1.26	0.30	1.01	1.83	0.71	0.11	0.07	0.02

6.3.5. Characterisation of the landscape fragmentation and configuration

In general, NP and PD increased for forests (Table 6.6 a), woodlands (Table 6.6 b) and savannahs (Table 6.6 c) from 1972 to 2014 in both PA and UPA. Over time, forest-LPI in PA increased markedly while an inverse trend was observed in UPA. This indicated that forests recovered in PA while experiencing fragmentation in UPA, corroborated by the trend of COHESION index. This could indicate that forest cover expanded in PA from cumulative areas of the increasing small patches in the landscape. Meanwhile, NP and PD for woodlands decreased from 1987 to 2014 in UPA whereas they increased for savannah from 1972 to 1987 and decreased constantly from 1987 to 2014. For all the periods, there was decreasing LPI and COHESION for woodlands in both PA and UPA. This could be due either to the conversion of woodlands into forests in PA or their fragmentation in UPA. This loss of woodlands caused the expansion of savannahs which LPI and COHESION increased over time in both PA and UPA.

Table 6.6a. Fragstat-based landscape indices for forest

Year	Status	NP	PD	LPI	COH	AI
1972	PA	55	02.20	00.28	78.31	69.96
	UPA	05	00.20	00.09	71.80	75.00
1987	PA	148	05.89	02.30	91.71	59.32
	UPA	27	01.08	00.05	51.83	36.23
2000	PA	177	07.09	03.72	93.47	70.44
	UPA	18	00.72	00.01	27.93	19.15
2014	PA	338	13.47	04.01	93.84	69.19
	UPA	62	02.47	00.37	73.45	53.77
References		NP ≥ 1	PD > 0	0 < LPI ≤ 100	0 < COH ≤ 100	0 < AI ≤ 100

Note: NP= number of patch; PD= patch density; LPI= largest patch index; COH= Patch cohesion; AI= Aggregation index; PA= Protected areas; UPA= Unprotected areas.

Table 6.6 b Fragstat-based landscape indices for woodland

Year	Status	NP	PD	LPI	COH	AI
1972	PA	28	01.12	47.24	99.59	87.19
	UPA	56	02.24	23.47	98.59	84.85
1987	PA	245	09.76	36.96	99.38	68.78
	UPA	787	31.35	05.61	92.72	56.09
2000	PA	296	11.86	28.16	98.56	75.43
	UPA	368	14.66	02.37	90.14	62.15
2014	PA	478	19.04	03.91	89.46	63.33
	UPA	99	03.94	0.06	50.19	40.07
References		NP ≥ 1	PD > 0	0 < LPI ≤ 100	0 < COH ≤ 100	0 < AI ≤ 100

Note: NP= number of patch; PD= patch density; LPI= largest patch index; COH= Patch cohesion; AI= Aggregation index; PA= Protected areas; UPA= Unprotected areas.

Table 6.6 c Fragstat-based landscape indices for savannah

Year	Status	NP	PD	LPI	COH	AI
1972	PA	75	03.01	24.43	98.51	85.60
	UPA	36	01.44	55.18	99.72	88.38
1987	PA	298	11.87	26.61	98.45	76.70
	UPA	98	03.90	61.11	99.79	73.75
2000	PA	257	10.30	25.52	95.06	71.86
	UPA	40	01.59	65.84	99.84	82.34
2014	PA	135	05.38	37.22	99.07	81.97
	UPA	40	01.59	63.27	99.76	88.92
References		NP ≥ 1	PD > 0	0 < LPI ≤ 100	0 < COH ≤ 100	0 < AI ≤ 100

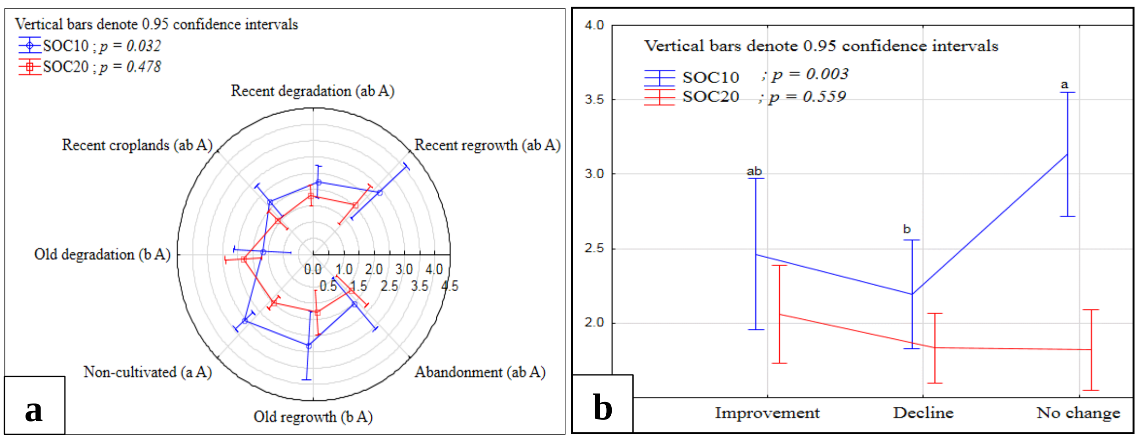
Note: NP= number of patch; PD= patch density; LPI= largest patch index; COH= Patch cohesion; AI= Aggregation index; PA= Protected areas; UPA= Unprotected areas.

6.3.6. Impacts of LUCC trajectories on soil ESS

a) Impacts on SOC contents at the sampling sites

The average SOC in the topsoil significantly differed ($p < 0.05$) under the different land conversion processes (Figure 6.4a). The topsoil of non-cultivated lands contained more SOC (3.06 %) whereas old degraded sites exhibited the lowest SOC (1.69 %). SOC in regrowth lands (old and recent) were quite high compared to those lands under recent cultivation and degradation. Though the highest value of mean SOC was recorded in old degraded lands (2.31 %), and the lowest in recent croplands (1.59 %), land conversion trajectories did not significantly affected subsoil SOC. These findings indicated that land conversion processes significantly affected the topsoil SOC, with less effect in the subsoil.

Further analyses according to the land performance categories indicated that no change areas exhibited the highest SOC (Figure 6.4b). However, the average subsoil SOC did not vary significantly according to the performance category, although non-cultivated lands still had the highest average SOC (1.99 %). Significant negative effects of vegetation decline on SOC were perceived at site level, especially for the topsoil.

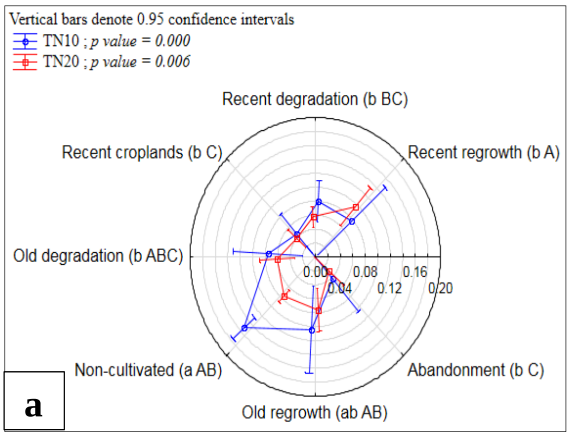


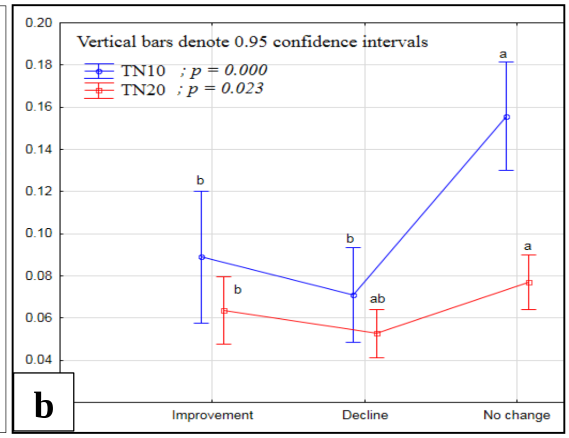
Note: Lower and capital letters indicate statistical difference for SOC10 (SOC for 0 - 10 cm) and SOC20 (SOC for 10 - 30 cm), respectively.

Figure 6.4 SOC according to LUCC trajectories (a) and performance (b)

b) Impacts on soil total nitrogen at the sampling sites

Similar to SOC, TN content under the various trajectories was significantly different for both topsoil and subsoil (Figure 6.5a). In the topsoil, non-cultivated (0.153 %) and old regrowth (0.104 %) lands had higher TN compared to abandoned lands (0.042 %) and lands under recent cultivation (0.045 %). Meanwhile, the subsoil TN were the highest in recent regrowth areas (0.097 %). Similar to the topsoil, TN in subsoils of abandoned and recent cultivated lands exhibited the lowest average values of 0.03 % and 0.04 %, respectively. In aggregate, non-cultivated lands had the highest TN content (0.229 %) whereas abandoned lands (0.072 %) and recent croplands (0.085 %) exhibited the lowest average values. Meanwhile, unchanged vegetation was the performance category with the richest TN in all layers (Figure 6.5b). In general, land conversion processes significantly affected TN up to 30 cm depth i.e. vegetation decline areas exhibited the lowest TN content, indicating the negative effects of vegetation decline on TN up to 30 cm depth.





Note: Lower and capital letters indicate statistical difference for TN10 (for 0 - 10 cm) and TN20 (for 10 - 30 cm), respectively.

Figure 6.5 Soil TN according to LUCC trajectories (a) land performance (b)

c) Impacts of LUCC trajectories on potential soil loss at the sampling sites

Table 6.7 shows that the average gross soil loss (GSL) under the different LUCC trajectories showed a significant difference, with recent vegetation regrowth areas indicating highest erosion prevalence (398 Mg $ha^{-1}y^{-1}$). About 20 to 21 Mg $ha^{-1}y^{-1}$ was modelled in abandoned and old degraded lands. The areas under recent cultivation exhibited the second highest eroded soil (about 73 Mg $ha^{-1}y^{-1}$). In general, soil erosion patterns were erratic and driven by site characteristics, with a wide range of values the LUCC trajectories as indicated by the large standard deviations. This could be due to the heterogeneous conditions of the investigated sites. GSL at the site level did not show any significant difference for the land performance categories. Areas with vegetation recovery (99 Mg.ha^{-}

$^{1}y^{-1}$) and decline areas (52 Mg.ha^{-1}y^{-1}) were highly vulnerable to soil erosion. This situation can be explained by the fact that the vegetation recovery areas were mostly areas with high human pressures that induced loss of vegetation protective cover.

Table 6.7. Potential effects of LUCC trajectories on soil erosion

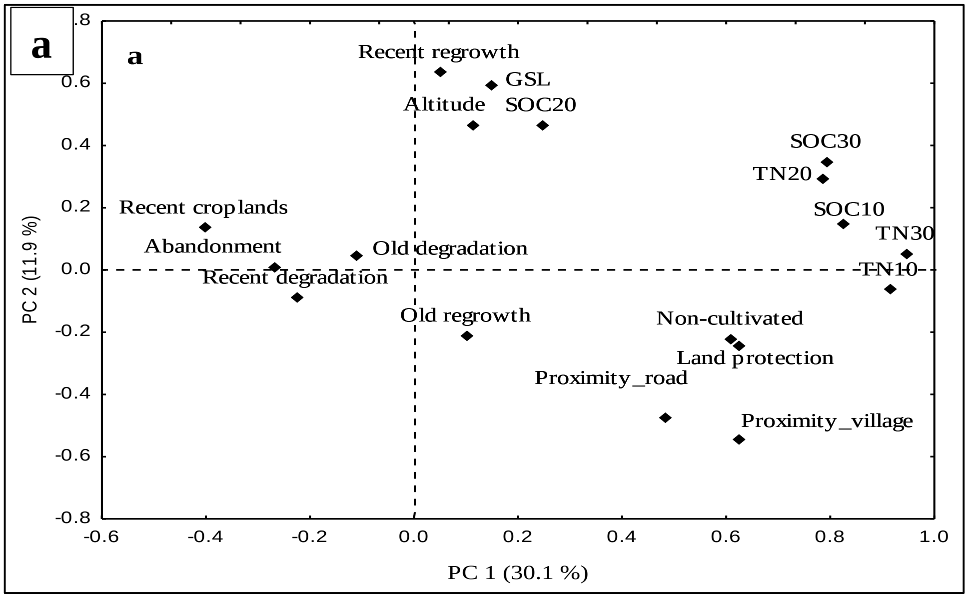
		Gross soil loss (Mg ha^{-1}y^{-1})
Level	Cases	Mean (StDev) **
Abandonment	4	19.51 (9.98) b
Non-cultivated	30	46.07 (54.32) b
Old degradation	4	20.80 (30.96) b
Old regrowth	11	60.07 (58.61) b
Recent croplands	18	73.13 (154.68) b
Recent degradation	3	44.30 (39.87) b
Recent regrowth	5	398.33 (341.53) a
Quality		Mean (StDev) ns
Improvement	34	98.9 (190)
Decline	11	51.5 (93.3)
No change	30	48.8 (58.3)

Note: Single and double stars indicate significant difference at 95 % and 99 % CI. Lower case letters indicate statistical difference. ns = non-significant. N = number of representative samples.

6.3.7. Interactions between environmental variables and soil conditions

In relation with environmental variables prevailing at site level (Table 6.8), soil erosion seemed to be more induced by landform through the high correlation with most of the topographic indices, especially slope and LS factor (≥ 0.74) (Table 6.9). Among all RUSLE factors, only LS was well correlated with GSL. In relation with other topographic indices, high altitudes were often associated with high erosion potential on steep slopes, as indicated by the positive correlation Alt, Slope and Alt.ch. Soil erodibility (K) did not show any significant influence on the GSL patterns. The huge GSL under recent vegetation regrowth could be explained by the high values of Slope, Alt.ch and LS factor occurring at these sites located close to riverbanks and high altitudes. Thus, GSL appeared to be more induced by topography than land conversion processes. This was evidenced by high GSL in PA, which often fall on steeper lands and healthy vegetation cover (positive correlation of 0.28

between LS and land management regime versus negative correlation of -0.39 between Land_Man and C factor).



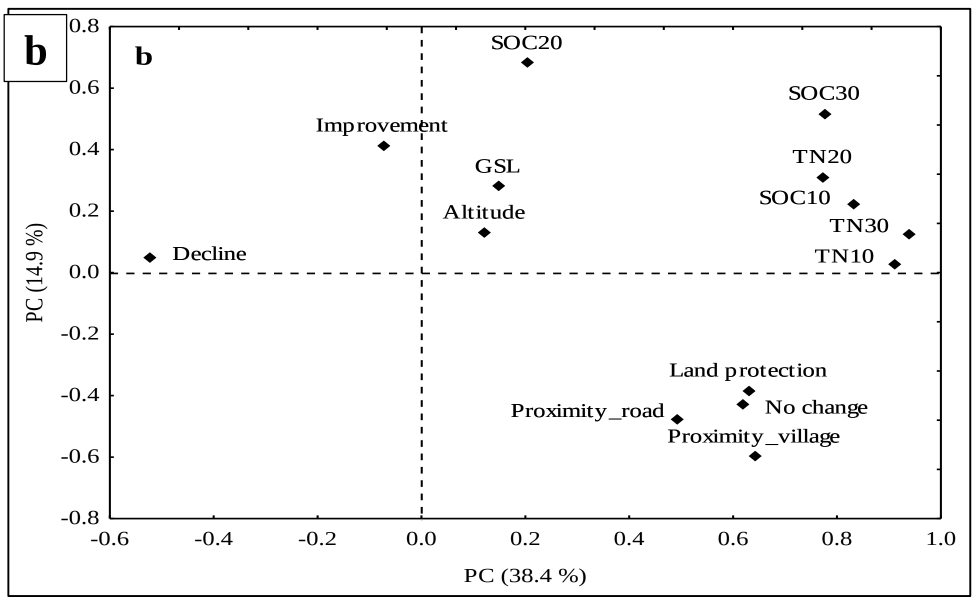


Figure 6.6. PCA plots of the effects of LUCC trajectories (a) and land performance categories (b) in relation with biophysical parameters on soil conditions

As a surrogate of soil erosion impact on soil nutrients, there was no clear interaction pattern. However, SOC and TN in upper 10 cm exhibited significant positive correlation (0.25 and 0.24) with the GSL (Table 6.7 and Figure 6.6), indicating that high amounts of sediment yield tend to increase soil nutrient at deposition sites. With regard to RUSLE factors, SOC and TN are negatively affected by R and C factors, indicating that erosivity associated with poor surface cover induced the depletion of SOC and TN, except subsoil SOC. The greater the R, the less the SOC and TN, indicating the detachment and transport of these soil parameters towards deposition sites. Similarly, the greater the C, the less the SOC and TN content. Among other environmental factors, proximity variables (distances to village and road) and land protection status showed significant positive correlation with soil nutrient in topsoil SOC and TN, and TN in the subsoil and the overall depth. This suggests that soils are nutrient-richer for further distances from roads and village centres, which distances mostly fall in core PA (Figure 6.6a and b). The distribution of SOC and TN is affected by LUCC, with non-cultivated lands being positively correlated with those soil parameters. However, SOC20 shows high prevalence in highlands, erosion-prone and recent regrowth areas. Land transformation inducing vegetation loss such as deforestation and degradation as well as land abandonment exhibited negative correlations with soil parameters, suggesting that a decline of land cover quality negatively affects soil conditions (Figure 6.6b).

Table 6.8. Summary of the environmental variables at site level

	K fctor	R factor	C factor	LS factor	SPI	TWI	Alt.ch	Slope	D_village	D_road	Land_man
Level Anova	Mean(StDev) ns	Mean (StDev) ns	Mean (StDev) **	Mean (StDev) **	Mean (StDev) ns	Mean (StDev) ns	Mean (StDev) ns	Mean (StDev) **	Mean (StDev) **	Mean (StDev) *	Mean(StDev) **
Abandonment	0.096 (0.000)	733.28 (5.19)	0.47 (0.17) a	1.16 (0.91) b	342 (276)	11.28 (0.38)	15.83 (2.69)	0.06 (0.03) b	873 (619) b	1008 (584) b	0.00 (0.00) c
Non-cultivated	0.092 (0.017)	715.38 (25.07)	0.32 (0.04) c	2.60 (3.12) b	237102 (1273262)	12.14 (2.86)	17.46 (22.16)	0.09 (0.06) b	5766 (4193) a	5347 (3785) a	0.57 (0.50) ab
Old degradation	0.096 (0.000)	729.44 (24.19)	0.37 (0.04) b	1.07 (1.36) b	320 (295)	11.58 (2.04)	14.73 (17.87)	0.06 (0.06) b	4198 (4520) ab	3916 (4135) ab	0.20 (0.45) bc
Old regrowth	0.096 (0.000)	714.28 (29.18)	0.36 (0.04) bc	4.53 (7.08) b	632 (846)	10.71 (1.62)	13.71 (9.26)	0.11 (0.11) b	7073 (4210) a	6638 (4208) a	0.75 (0.50) a
Recent croplands	0.096 (0.000)	735.06 (6.69)	0.44 (0.05) a	2.61 (6.19) b	2839 (5885)	12.32 (2.12)	7.37 (6.34)	0.07 (0.09) b	1348 (618) b	2145 (1620) b	0.00 (0.00) c
Recent degradation	0.111 (0.036)	721.9 (32.35)	0.37 (0.05) b	2.28 (1.79) b	970 (1361)	11.00 (1.39)	22.74 (30.5)	0.09 (0.04) b	3073 (1719) b	4524 (3783) ab	0.22 (0.43) bc
Recent regrowth	0.096 (0.00)	735.59 (5.32)	0.28 (0.08) c	24.92 (20.06) a	3256 (3025)	10.08 (0.86)	46.45 (37.48)	0.35 (0.28) a	1705 (106) b	658 (626) b	0.67 (0.58) ab
Quality	Mean (StDev) *	Mean (StDev) **	Mean (StDev) **	Mean (StDev) ns	Mean (StDev) ns	Mean (StDev) ns	Mean (StDev) ns	Mean (StDev) ns	Mean (StDev) **	Mean (StDev) **	Mean (StDev) **
Improvement	0.104 (0.028) a	726.52 (25.81) a	0.39 (0.07) a	2.27 (3.75)	1514 (3573)	11.44 (1.74)	17.55 (24.34)	0.08 (0.06)	2594 (2270) b	3660 (3372) b	0.15 (0.36) b
Decline	0.096 (0.000) b	732.01 (16.87) a	0.36 (0.11) ab	6.22 (11.91)	411348 (1692150)	11.58 (3.10)	17.2 (20.35)	0.12 (0.16)	3153 (3063) b	2682 (3240) b	0.35 (0.49) ab
No change	0.092 (0.019) b	710.65 (24.83) b	0.32 (0.04) b	2.79 (3.37)	5367 (20514)	11.97 (2.21)	18.94 (23.65)	0.09 (0.07)	6472 (4253) a	6048 (3724) a	0.64 (0.49) a

Note: Single and double stars indicate significant difference at 95 % and 99 % CI whereas ns indicates not significant. Lower case letters indicate post-hoc statistical difference. K, R, LS and C factors are the input parameters of the RUSLE model. SPI = stream power index; TWI = topographic wetness index, Alt.ch = altitude above channel; D_road = distance to the main road; D_village = distance to a village center; Land_man = Land management regime.

Table 6.9. Correlation between soil parameters and environmental factors

	SOC10	TN10	SOC20	TN20	SOC30	TN30	K factor	R factor	SPI	TWI	Alt.ch	Slope	D_village	D_road	Land_man	GSL	C factor
TN10	**0.66***	1.00															
SOC20	0.17	0.20	1.00														
TN20	**0.59***	**0.66***	**0.41***	1.00													
SOC30	**0.86***	**0.64***	**0.62***	**0.67***	1.00												
TN30	**0.73***	**0.94***	**0.31***	**0.85***	**0.73***	1.00											
K factor	-0.18	-0.13	-0.18	-0.17	-0.18	-0.17	1.00										
R factor	**-0.42***	**-0.34***	0.20	-0.15	**-0.24***	**-0.28***	0.06	1.00									
SPI	**0.30***	**0.35***	0.10	0.21	**0.30***	**0.35***	-0.04	0.00	1.00								
TWI	-0.01	0.04	0.06	-0.03	0.03	-0.01	-0.01	0.04	**0.41***	1.00							
Alt.ch	0.18	0.15	-0.05	0.21	0.12	0.20	-0.11	-0.12	0.15	**-0.40***	1.00						
Slope	**0.24***	**0.24***	0.00	0.18	0.17	**0.26***	-0.05	-0.01	**0.44***	**-0.53***	**0.48***	1.00					
D_village	**0.32***	**0.56***	-0.09	**0.43***	**0.23***	**0.56***	0.06	**-0.51***	0.10	-0.17	0.12	0.19	1.00				
D_road	**0.30***	**0.43***	-0.11	**0.33***	0.20	**0.40***	0.05	**-0.65***	0.08	-0.10	0.09	0.12	**0.72***	1.00			
Land_man	**0.35***	**0.48***	-0.06	**0.37***	**0.27***	**0.50***	0.16	**-0.38***	0.11	-0.21	0.15	**0.26***	**0.60***	**0.33***	1.00		
GSL	**0.25***	**0.24***	0.12	0.15	**0.26***	**0.25***	0.18	0.04	**0.65***	-0.16	**0.27***	**0.74***	0.10	0.08	0.20	1.00	
C factor	**-0.25***	**-0.56***	-0.11	**-0.44***	**-0.26***	**-0.56***	0.13	0.19	**-0.26***	0.16	**-0.25***	**-0.30***	**-0.36**	**-0.28***	**-0.39***	-0.09	1.00
LS	**0.43***	**0.43***	0.07	**0.30***	**0.36***	**0.43***	0.02	-0.05	**0.60***	**-0.25***	**0.38***	**0.81***	0.22	0.13	**0.28***	**0.78***	**-0.31***

Note: Correlation values are stared significant at 95 % CI. TN10, TN20 *and* TN30 *stand for TN in 0 - 10 cm, 10 - 30 cm, and 0 -30 cm, respectively. The same nomenclature applies to SOC.* K, R, LS *and* C *factors are the input parameters of the RUSLE model.* SPI = *stream power index;* TWI = *topographic wetness index,* Alt.ch = *altitude above channel;* D_road = *distance to the main road;* D_village = *distance to a village center;* Land_man = *Land management regime;* GSL = *gross soil loss.*

6.4. Discussion

6.4.1. LUCC trajectories and soil ESS

During the overall period, the high rates of natural vegetation degradation (0.9 %) and agricultural deforestation (0.3 %) showed that a great part of the Mo basin experienced negative land transformations due to an increasing human appropriation of the landscapes. This conversion of native lands to less vegetation cover ones such as agricultural fields, significantly decreased SOC and TN, especially in the topsoil. In line with findings in savannah areas of Togo, reference natural forests or undisturbed lands stored more SOC than the surrounding lands converted into croplands (Sebastia et al., 2008). As reported by many similar studies (Kintché et al., 2010; Lal, 1993; Traoré et al., 2015; Wiesmeier et al., 2013a), the observed decrease of SOC and TN following the vegetation decline could be due to the subsequent reduction of OM availability from litter and micro-organisms. Though the magnitude of SOC and TN loss due to land cover decline is not characterised for specific LUC types, the results from this study gave insight to the understanding of soil nutrient loss due to land conversion, especially forest degradation and agricultural deforestation. Furthermore, such LUCC could also affect the surface hydrological processes at watershed level, especially soil erosion by water. Several studies indicated that soil erosion influences the nutrient balance in on-site and off-site soils (Guillaume et al., 2015; Lacoste et al., 2015; Mondini et al., 2012). Considering the on-site effects, soil loss had poor agreement with SOC and TN contents, though it is expected that erosion and transportation induce a decline in on-site soil quality (Lal, 2014).

In addition, many other factors such as land use history, soil types, topographic and climatic conditions should be accounted for the full spectrum of drivers of SOC and TN behaviour at the landscape or site levels (Lal, 2014; Liu et al., 2011; Traoré et al., 2015;

Wiesmeier et al., 2014b). The leaching effects by the surface runoff associated with the mountainous conditions and the LUCC processes at the different topographic positions might lead to such observations in the Mo basin. However, the correlation between soil loss and the contributing factors (i.e. RUSLE factors) showed that LS factor seriously contributes to the soil erosion patterns in the landscape. This situation is similar to the findings of Tamene et al. (2006) who reported that landform, in combination with LUCC, is the most critical component controlling soil erosion in the mountainous areas. In this regard, it is suggested that best practices that control and protect both slope and vegetation cover will reduce soil loss potential.

With regard to the landscape fragmentation and potential effects on ESS was analysed through LM. The increase over time in NP and PD is an indicator of the decrease of landscape homogeneity and integrity across the river basin. Further, LPI, COH and AI decreased constantly over time indicating an increasing loss of landscape and habitat connectivity. Similar trends observed in LM for both PA and UPA demonstrate the weak level of law enforcement regarding the PA in the river basin (Wala et al., 2012), though natural processes had been mentioned as drivers of vegetation decline in West African environments (Le et al., 2012b; Traore et al., 2015). Mazgajski et al. (2010) also noted that the process of forest and landscape fragmentation consists of both habitat loss and change in in the provision of landscape services. In this regard, Shi et al. (2013) reported that the increase in NP of LUC types might significantly accelerate soil erosion and increase sediment export. The same authors reported AI, COHESION, LPI to be metrics controlling the watershed soil erosion patterns. These metrics reflect the physical connectedness and aggregation of land cover types within landscapes, and have higher values when LUC types are more clumped and aggregated (McGarigal et al., 2012). It is concluded that the observed increasing trends of AI, COHESION and LPI in the Mo basin indicated landscape fragmentation, which might induce loss of proper provision of ESS. Further, proximity analyses showed that areas close to roads and villages, and gentle landscapes experienced more fragmentation. This indicated that the land use decision-making and land allocation in the study area are based on easy access conditions of lands.

6.4.2. Implications for management of landscapes and ESS

With regard to the change trends in LUC and the processes involved, land management and conservation is revealed to remain a challenge in Mo basin. The ongoing decline in natural vegetation cover, even in PA is an evidence of such land decline. However, the relative low rates of occurrence compared to the situation in UPA are indicators of the importance of PA in limiting the level of human encroachments. Processes involving land cover improvement and natural vegetation cover regrowth were predominant in PA for all transition periods. Nevertheless, land improvement was also significantly observed in certain parts of UPA, especially in access-limited areas by distance or rugged topography conditions. On the other hand, the interactions between soil loss and environmental variables indicated that GSL is more induced by topography rather than land conversion processes. In this regard, it will be valuable that land management options concentrate efforts towards the protection of such sensitive areas commonly located at higher slopes of the landscape (Tamene et al., 2014; Tamene et al., 2006). Regarding the efficient management of soil nutrients, soil erosion control as well as land cover monitoring for DFD reduction will have significant effects on SOC and TN contents in the basin. In line with the correlation outputs, soil erosion is often associated with low soil nutrients, exception made for subsoil TN. This situation indicates that abating soil loss and DFD could significantly reduce the loss of SOC and TN concentration up to 30 cm.

6.5. Conclusion

This study assessed the effects of LUCC trajectories on landscape patterns, SOC, TN, and soil erosion potential in the Mo River basin. The clustering of the LUCC into conversion categories and trajectories showed that non-cultivated lands and vegetation regrowth areas were widely dominant in the landscapes during all the periods. However, the proportions of these trajectories declined progressively with the emergence of the human-transformed lands. LUCC trajectories occurred at different magnitudes irrespective to the land protection regime, inducing landscape fragmentation, and subsequent loss of proper provision of ESS, especially in the areas close to roads and villages, and gentle landscapes experienced more fragmentation. The study also revealed that SOC and TN were significantly affected by LUCC trajectories. Land conversion inducing vegetation cover decline affect negatively SOC and TN, especially in the topsoil. However, LUCC trajectories marginally affected soil loss patterns at site specific level, which are more likely controlled by landforms. As undisturbed or unchanged natural vegetation served as reference line, the differences in soil conditions (SOC, TN and soil loss) of the different LUCC categories were a good indication of the effects of LUC history on the fields. In the absence of historical data for chronosequence analysis, the approach used in this study helped in capturing the potential impacts of landscape change on ESS in the Mo basin. The findings of this study improve our knowledge of the impacts of LUCC on soil-landscape conditions in the area. They also provided a basis to design spatially explicit tools, formulating and implementing management strategies for integrated landscapes.

CHAPTER 7: SIMULATION OF SOIL EROSION TO SUPPORT SUSTAINABLE LAND MANAGEMENT IN THE MO BASIN [5]

7.1. Introduction

All over the world, land degradation (LD) through soil loss by water is one of the oldest serious environmental challenges affecting land productivity and water resources. Soil is a component of the landscape system, a vital resource for producing services to support an increasing world population (Rhodes, 2014). Thus, soil quality has gained interest, as it is at the forefront of issues relating to environmental monitoring and food security (Oladele & Braimoh, 2011; Stockmann et al., 2015). According to the Millennium Ecosystem Assessment, the rapid degradation of soils and water seriously compromises ESS in multifunctional landscapes and reduces the resilience of food security (MEA, 2005). Since soil is critical for sustainable landscapes and food security, monitoring its quality is important to predict and anticipate protective measures.

Processes of soil quality loss often include soil erosion by water, which occurs at diverse severity levels depending on the contributing factors that include natural processes and land management practices (Tamene et al., 2006; Vlek et al., 2008). Hydric-induced soil loss is understood as soil particles depletion due to water effects through surface runoff, rill and inter-rill, and gully (Martin-Fernandez & Martinez-Nunez, 2011; Shoshany et al., 2013). Surface or sheet and rill erosion refer to processes of soil particles detachment and transport by surface runoff while gullies are surface overland flow occurring through permanent flow channels. In any condition, net soil losses occur when erosion rates are

[5] This chapter is prepared as manuscript to be submitted to *Earth Surface Processes and Landforms*.

greater than deposition or soil formation rates and the tolerable soil loss varies according to regions and land use and management purposes. Therefore, the important factors influencing hydrological processes (e.g. soil erosion patterns within a catchment) need to be monitored to ensure sustainable landscape management.

Modelling soil loss has supplanted the traditional time consuming methods of soil monitoring with regard to long-term perspectives and many other spatial considerations. Considering soil erosion as a spatio-temporal phenomenon, it is represented through methods employing equations describing the link between environmental parameters that offer better explanations of the phenomenon occurrence (Rhodes, 2014). Thus, depending on the data availability, soil erosion is commonly modelled using different models. Among the models, the low input parameters and its easier implementation in various environments enhance the selection of the RUSLE and derivatives to model soil erosion (Ashiagbor et al., 2013; Fathizad et al., 2014; Guo et al., 2013; Kim, 2006; Le et al., 2012b; Owusu, 2012; Tamene et al., 2014; Zhou et al., 2014c). The RUSLE model has the capability to be implemented in GIS environment and be coupled to other spatial explicit models to represent soil erosion. For instance, the Landscape Planning and Management Tool (LAPMAT) is a spatially distributed model built on RUSLE and Sediment Delivery Ratio (SDR) (Tamene and Le, 2014). Hence, spatially explicit models (SEM) such as LAPMAT allow the understanding of the contribution of causal factors of the phenomenon being modelled (Le et al., 2012a; Verburg et al., 2013).

Since pro-active approaches accounting for efforts to assist conservation of land ecosystems are necessary, the implementation of SEM is important to foster the understanding of the complexity of the factors and actors behind the degradation processes. In this regard, it is judicious to implement a SEM to the case landscape of the

Mo basin where the extent of soil erosion has not yet been widely quantified at the catchment scale. It is proposed to assess the soil erosion dynamic in relation with LUCC and landforms using a clone version of the LAPMAT adapted to the Mo basin. This tool for supporting land management in mountainous areas is to address the crucial local challenges of LD. Therefore, an adapted version of LAPMAT, the LAMPT_Mo (LAndscape Management and Planning Tool for the Mo basin), is used to simulate historical soil loss from 1973 to 2014, identify erosion patterns and hotspots, and evaluate the efficiency of some possible soil conservation scenarios. The novelty of this SEM is its capabilities to integrate soil erosion dynamic in relation with LUC types and landform features, and generate timely spatially explicit information. The objective of this study was to model the soil erosion patterns and land management options for LD mitigation and landscape restoration in the Mo River basin.

7.2. Materials and Methods

7.2.1. Description of LAMPT_Mo and its specificities

The LAMPT_Mo is a clone version of the LAPMAT adapted to the Mo River watershed (Chapters 3 to 6 for details on the study area). Similar to LAPMAT, the clone model is implemented in a tropical mountainous region. It uses the programming framework NetLogo v5.2 (Wilensky, 1999) to adapt the sub-models. It integrates general features of the landscape (LUC units, landform parameters, soil types, and land use and management options) and rainfall data to simulate gross soil loss, sediment delivery ratio and the net sediment yield. Contrary to the original model, LAMPT_Mo is calibrated to a multifunctional landscape including large protected areas and agricultural landscapes. It also has the particularity of integrating historical LUC data relevant for evaluating soil erosion response to historical LUCC. In addition, LAMPT_Mo builds a default land management-supporting factor (P factor in RUSLE) based on a layer of PA network of the Mo river basin. In term of outputs, the clone model presents soil loss amounts according to land management regime (PA vs UPA), LUC types (only forests, woodlands, savannahs, and croplands) and the buffer zones along river/streams.

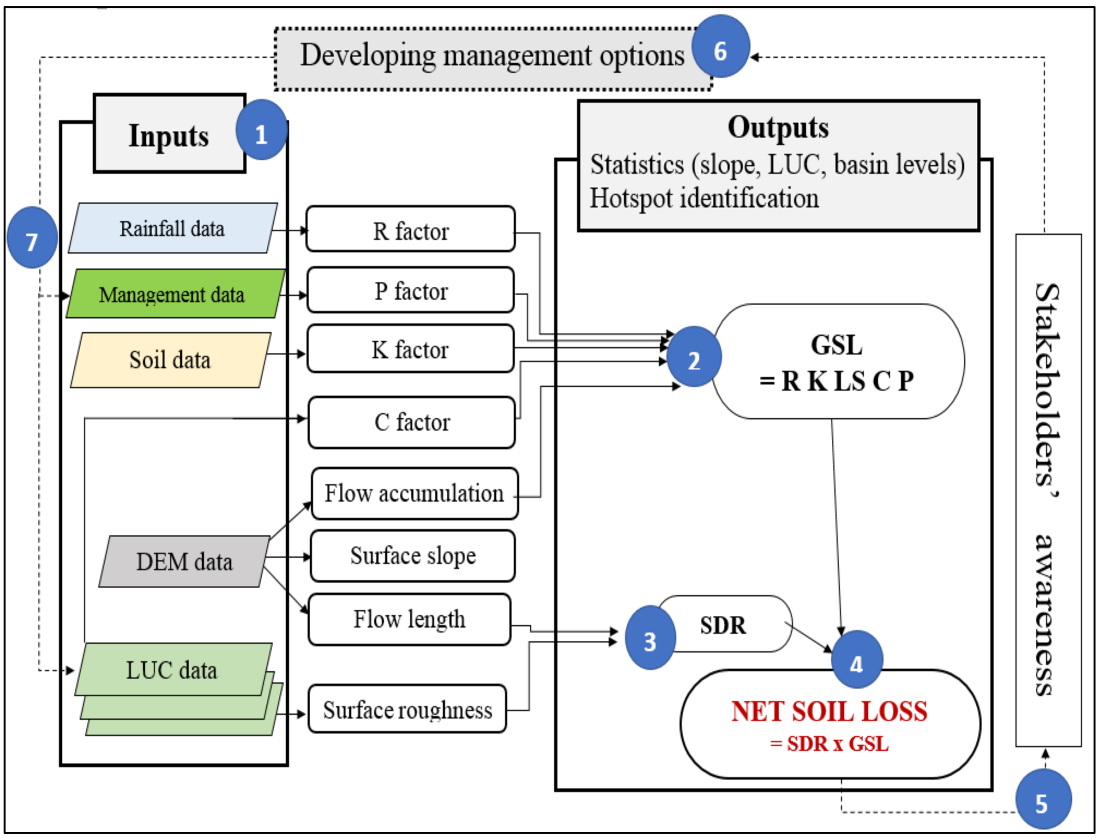


Figure 7.1. Process of soil loss modelling in the LAMPT_Mo

(*Source: author development based on Tamene et al. (2014)*).

Soil erosion modelling in LAMPT_Mo is fundamentally based on RUSLE model (Renard et al., 1997) described in Chapter 6 (Equation 6.1). In addition, a sediment delivery ratio sub-model (Equation 7.4) is used to represent the spatial patterns of soil erosion since RUSLE does not accommodate spatial dimension. The steps and major sub-models of the LAMPT_Mo are provided in Figure 7.1.

Figure 7.2 presents the graphical use interface of LAMPT_Mo adapted from the original LAPMAT (Tamene et al., 2014). Labelled feature 1 composes of options for importing input variables spatially explicated through maps (individually displayed via Feature 2). The selection of adequate values for mean annual rainfall, slope steepness and length, and definition of threshold constants for potential location of gullies (Moore et al., 1991) are performed under Feature 1. Management options for enclosing gully lands, erosion hotspot areas, and steep lands, as well as terracing steep lands and gully areas and updating P factor are proposed under the same Feature 1. Feature 3 offers the ability to compute the spatially explicit potential (STCI) and actual (GSL) soil loss, sediment delivery ratio (SDR) and net soil loss (NSL) at the landscape level. While the imported and generated variables as well as the simulated soil loss are visualised for their spatial configuration (Feature 4), graphical options are offered to visualise the outputs according to slope classes (Feature 5), land use/cover types and protection status (Feature 6), and river buffer zones (Feature 7). Features 8 and 9 are provided for exporting the outputs in tabular forms for usage in other analytical environment (GIS, statistical software, etc.).

The list of key inputs (Figure 7.3) are calibrated data for the study area in order to design real and suitable management options supporting land use and planning (Tamene et al., 2014). Different types of settings are relevant to design and implement LAMPT_Mo: Initial biophysical conditions (data on terrain and its derivatives, land use/cover, soil erodibility based on soil types, protection status of lands, buffer zones of river network, etc.) are prepared for the Mo River basin.

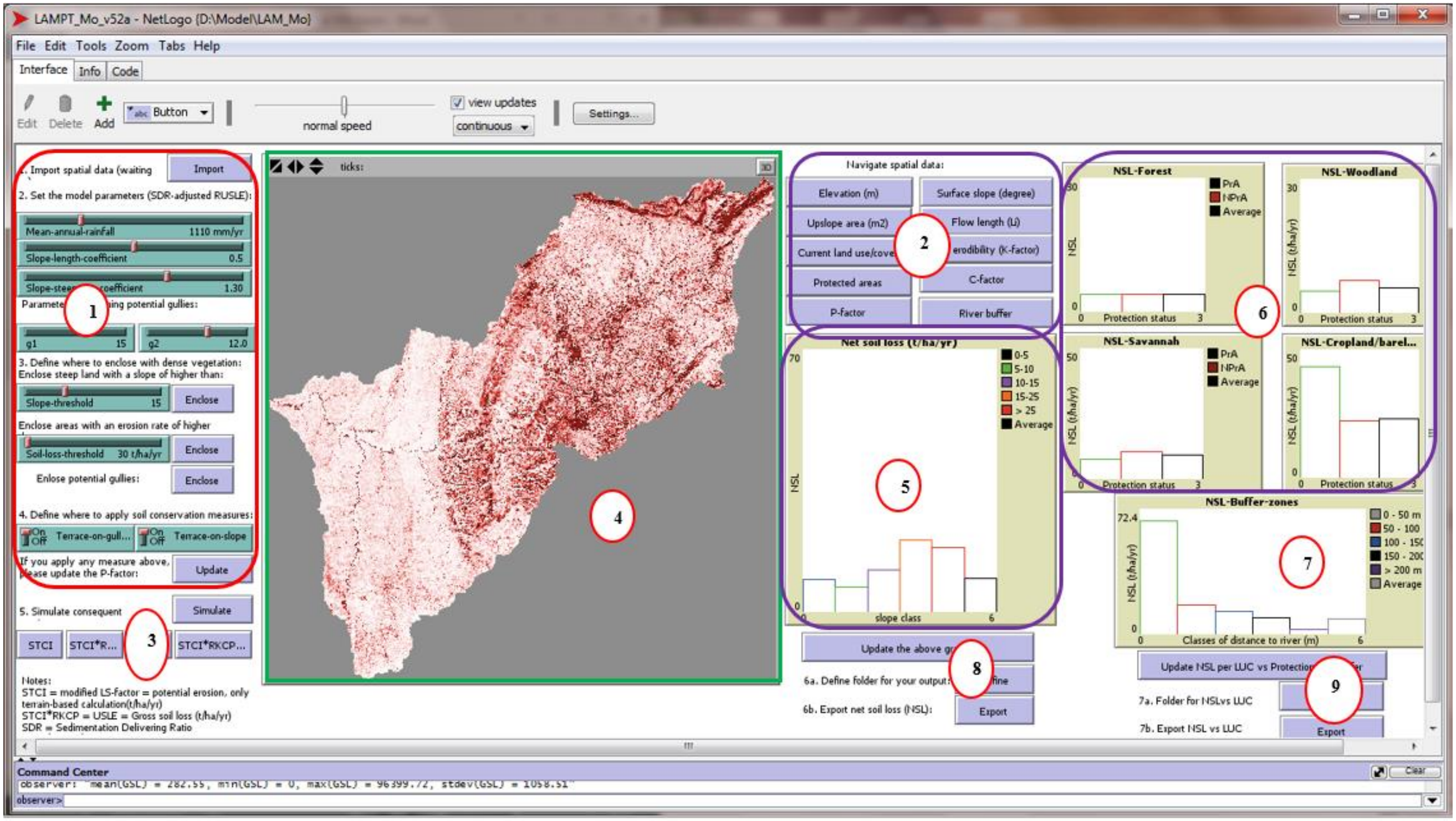


Figure 7.2. Graphical user interface of the LAMPT_Mo

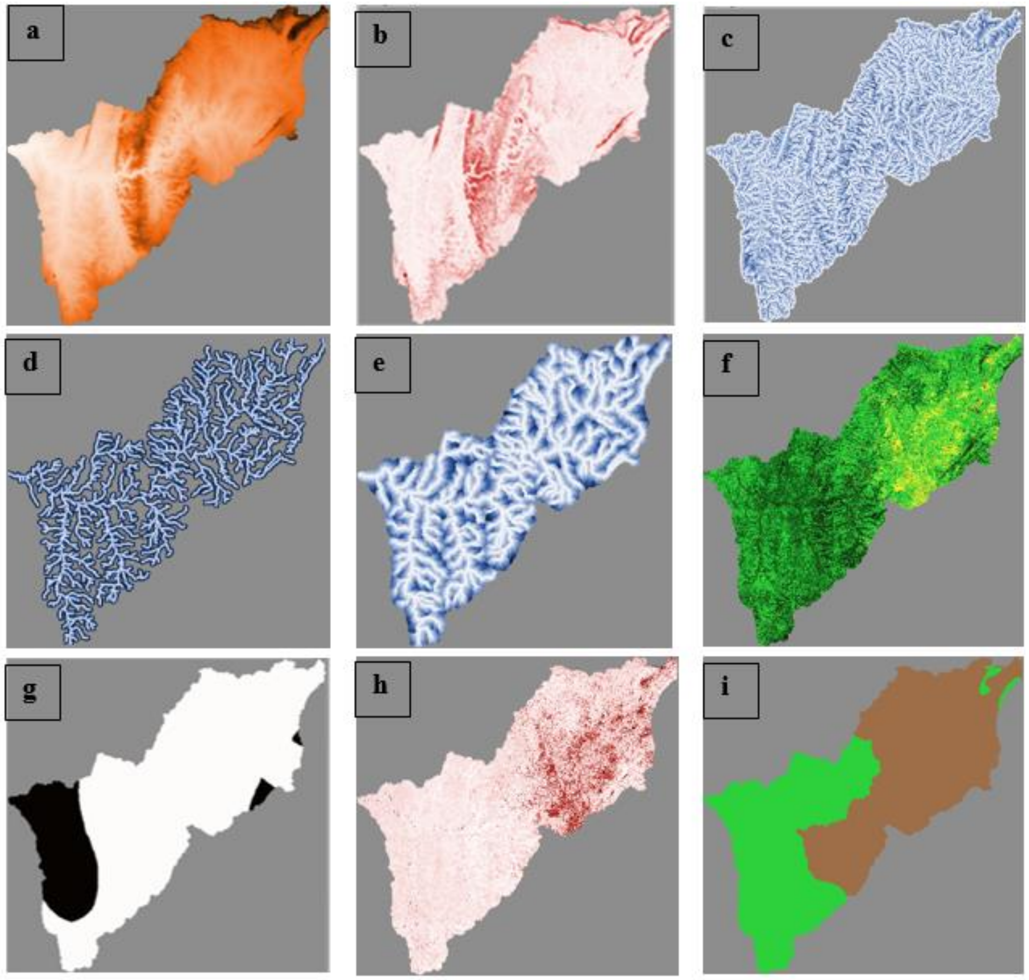


Figure 7.3. Input layers for LAMPT_Mo. Terrain elevation (a), Terrain slope (b), Surface flow accumulation (c), River buffer zones (d), Overland flow distance (e), land use/cover (f), soil erodibility factor (g), Soil cover factor (h), and protected lands (i).

7.2.2. Sub-models for estimating potential, gross and net soil losses

The potential soil loss risk is derived based on the Sediment Transport Capacity Index (STCI). STCI is a modified LS of the RUSLE model used to map the potential risk of soil loss at the landscape level. As a function of the upslope area ($\boldsymbol{As}$), the slope and its characteristics i.e. the slope-length ($\boldsymbol{\delta}$) and slope steepness ($\boldsymbol{\alpha}$) coefficients (Equation 7.1), STCI does not consider sediment deposition (Tamene, 2005). The particular interest of STCI resides in its capability to identify the most susceptible or vulnerable areas to soil erosion for developing conservation measures at the landscape level (Tamene, 2005).

$$STCI = \left(\frac{10A_s}{22.13}\right)^{\delta} \times (\delta + 1)\left(\left(\frac{\sin(slope)}{0.0896}\right)^{\alpha}\right) \qquad \text{(Equation 7.1)}$$

Next, the STCI is combined with other soil loss factors different from the LS, to predict the gross soil loss (GSL) and its spatial patterns in the Mo basin for the different periods of study and the different land management options at the landscape level. It aims at identifying the most erosion-prone areas, taking into account natural terrain and climatic conditions, as well as the human interferences on soil erosion susceptibility. However, as such, the predicted GSL (Equation 7.2) does not consider sediment deposition dimension.

$$GSL \ = \ STCI\ (KCPR) \qquad \text{(Equation 7.2)}$$

where ***GSL*** represents the potential long-term average annual soil loss (in t ha^{-1} yr^{-1}). ***R*** **(**MJ mm ha^{-1} h^{-1} yr^{-1}) is the rainfall and runoff factor by geographic location. The greater the intensity and duration of the rain storm, the higher the erosion potential. ***K*** is the soil erodibility factor (t ha h ha^{-1} MJ^{-1} mm^{-1}). ***C*** is a dimensionless factor for the soil cover-management. ***P*** is the dimensionless factor expressing the support practices of soil management such as terracing, stone lines, strip cropping, etc.

In order to take into consideration sediment deposition, LAMPT_Mo has been designed to estimate the net soil loss (NSL) from the gross soil loss. NSL_i at a pixel scale was computed based on the Equation 7.3 (Le et al., 2012; Tamene et al., 2014):

$$NSL_i \ = \ GSL \ \times \ SDR_i \qquad \text{(Equation 7.3)}$$

where *GSL* is the RUSLE amount of soil loss at a pixel size (in Mg $ha^{-1}y^{-1}$) and SDR_i is the sediment delivery ratio at pixel scale.

The Sediment Delivery Ratio (SDR) at pixel level (SDR_i) (Equation 7.4) is used as a sediment distributed model to indicate the probability that eroded particles mobilized from an individual cell are transported to the nearest stream pixel (Stefano et al., 2005)

$$SDR_i = \exp\left(-\beta * \frac{L_i}{R_i(S_i)^{1/2}}\right) \qquad \text{(Equation 7.4)}$$

where $\boldsymbol{\beta}$ is a routing coefficient set -0.0051; $\boldsymbol{L_i}$ is the length of segment i in the flow path and is derived from the length of the side or diagonal of a cell depending on the flow direction in the cell (in m). In the LAMPT_Mo, $\boldsymbol{L_i}$ is the flow length layer imported as initial input (See LAMPT_Mo description). $\boldsymbol{R_i}$ is the coefficient of surface roughness characteristics (m/s) derived based on the look-up table (Table 7.1). $\boldsymbol{S_i}$ is the slope gradient (m/m) generated from the surface slope (Equation 7.5).

$$S_i = \ tan\ (\alpha \ + \ 0.01) \qquad \text{(Equation 7.5)}$$

where $\boldsymbol{\alpha}$ is the surface slope (in ^{O}) derived from surface elevation model.

The roughness of the earth surface i.e. land surface cover (e.g., roads, ground) and the objects hereon (e.g., buildings, vegetation, etc.) are of important interest in terms of hydrodynamic friction (Dorn et al., 2014) and therefore affect water flow and sediment transport, and re-deposition on landscape under investigation (McCuen, 1998). Roughness reflects the influence of the surface on the momentum and energy dissipation in resisting the flow of the fluid (Galema, 2009). Manning's roughness coefficient is the most common parameter used to express surface roughness in surface hydrological processes (Kalyanapu et al., 2009; Tamene et al., 2014). Table 7.1 shows the selected appropriate ***Ri*** values for each LUC type in the Mo basin.

Table 7.1. Surface roughness coefficients for overland flow (Chow, 1959; Engman, 1986; McCuen, 2005; Tamene et al., 2014)

LUC types	***Ri***
Forests	0.35
Woodlands	0.76
Savannahs/shrubs (Grasslands)	1.56
Croplands	2.13

7.2.3. Development of other input parameters

***a) Erosivity factor* (R)**

As time-variant variable, the rainfall erosivity data (**R**) were generated from the annual rainfall of each year of study. The average annual rainfall were obtained from the Global Weather Data (http://globalweather.tamu.edu/) (Dile & Srinivasan, 2014; Fuka et al., 2014). The methodology for **R** factor calculation is provided in Chapter 3.

***b) Soil cover-management factor* (C)**

The soil cover-management factor (**C**) was derived from input layers informing on current and historical LUCC. Four LUC maps i.e. 1972, 1987, 2000 and 2014, were used in this study. Focus was given to four major LUC types: forests, woodlands, savannahs, croplands. The main reasons are their particular involvement in biogeochemical cycles and climate change mitigation as well as the agricultural land management for food security. For LUC definition, please refer to Chapter 4 and Chapter 5. For each LUC unit, **C** factor was derived and calibrated from the look-up Table 7.2. These values were used to generate a **C** factor map for each of individual four dates of study.

Table 7.2. Soil surface cover factor (C) for the LUC types (Tamene et al., 2014)

LUC types	C factor
Forests	0.001
Woodlands	0.01
Savannahs/shrubs (Grasslands)	0.025
Croplands (orchards/parks)	0.15

c) Management supporting practices factor (P)

Supporting management systems are not really observed at a large scale in the catchment as well as in the Volta basin (Le et al., 2012b; Tamene and Le, 2015). Therefore, **P** factor is equal to a default value of 1 in UPA and 0 in PA (Tamene, 2005). These initial **P** values are defined based on PA network of the Mo river basin. The influence of the supporting land management options on soil erosion mitigation is assessed via some provided options in LAMPT so that **P** factor could be updated depending on user preferences. The updating function is transferred into a **P** factor map based on the conditions defining the land management options according to the user preferences. For instance, enclosing gullies is defined as transforming those areas into woodlands with **P** value of 0.6.

d) Spatial data on terrain elevation and surface conditions

The elevation model data used in this model is derived from the USGS website (https://earthexplorer.usgs.gov.) as described in Chapters 3, 4 and 6. It served in deriving information on terrain conditions such as flow length (or overland flow distance), the slope length (LS factor), the upslope area (or flow accumulation), stream network, etc. These terrain inputs were set time –invariant even though some of the terrain attributes could change over time depending on surface hydrological processes erosion inducing river enlargement and bank collapse (Fang et al., 2013; Tamene & Vlek, 2007).

- **Flow length (L_i) and upslope contributing area (A_s)**

The flow length known as the distance from the centre of a cell to a stream or the watershed outlet. It has a significant impact on the spatial patterns of soil erosion and deposition. Accounting this parameter for the modelling of soil erosion is essential in order to capture the process of sediment transport and deposition in the watershed. Therefore, flow length data are derived from SRTM DEM (Chapters 3 and 4) following numerous steps in SAGA GIS environment. From the DEM, all sinks that could trap sediments and water flow through channels were filled to facilitate the hydrological modelling process of soil loss/transport (Planchon & Darboux, 2001). Upslope contributing area ($\boldsymbol{A_s}$) and flow length ($\boldsymbol{L_i}$) were calculated from the Multiple Flow Direction algorithm (Freeman, 1991) using SAGA software[6] (http://www.saga-gis.org/en/index.html) (Conrad, 1998).

- **Slope length and steepness factor (LS)**

In various landscapes, land surface and topography determine the patterns of hydrological processes such as soil erosion by water. The surface overland flow and runoff are determined by these surface conditions that affect hydrological connectivity of each point of the landscape (Moore et al., 1991; Reaney et al., 2014). The control and consideration of the slope length and steepness are the major considerations for mitigating land surface processes in mountainous areas. In RUSLE model, terrain effects are accounted by the LS factor that expresses the total sediment yield from each point of the landscape. According to Renard et al. (1997), soil loss is proportional to the increase in slope steepness with less sensitivity to slope length. In this study, the values of slope length and steepness coefficients (defining the LS factor for RUSLE) are set to be 0.50 and 1.30.

7.2.4. Description of scenarios and simulation outputs

According to Economics of LD initiative, scenario analysis or planning is a ‘structured process of exploring and evaluating alternative futures’, whose ultimate aim is to illustrate the consequences of policy options, inform and improve decisions. In this context, two

[6] SAGA= System for Automated Geoprocessing and Analyses (http://www.saga-gis.org/en/index.html) is the GIS package that was rooted from DiGEM, developed by the same author.

categorical scenarios were defined for simulating soil loss and propose alternatives for adapted land management: (i) simulation based on business as usual (BAU or S0) conditions, and (ii) simulation based on LUC management options (S1). For all scenarios, simulated soil loss is analysed to highlight the contribution of specific LUC types, slope-classes and river buffer zones to soil erosion. Most of the scenario designs result from LUC-reorganisation at the landscape level in order to identify viable options that significantly reduce soil loss (Tamene, 2005; Tamene et al., 2014). In all of the scenarios proposed in the LAMPT_Mo, default values of model variables are offered according to the conditions in the Mo river basin. Nevertheless, large ranges of values are provided for selecting appropriate design of conservation in specific sites and when facing uncertainties related to the selection of values (Tamene et al., 2014).

a) Scenario of business as usual for historical soil loss assessment

Simulation was performed for the status quo (BAU) i.e. annual soil loss/sediment yield rate and its spatial pattern for 2014 was calculated based on the landscape conditions representing the existing land conditions (Tamene, 2005; Tamene et al., 2014). The result of 2014 served as reference data for comparison of the simulated soil erosion/sediment yield of 1972, 1987 and 2000 in order to highlight not only the effects of LUCC but also the potential effects of changing rainfall. Since there is no historical reference for the Mo basin for quantitative validation of the simulated historical NSL, the reference simulation for 2014 was used to qualitatively validate the aforementioned data based on LUCC.

b) Management scenarios (S1): reducing soil loss at landscape level

Management options are built to express scenarios that focus on reorganising LUC types and adopting conservation practices across landscapes based on predefined criteria, such as conserving gullies and their buffer zones (S1G), managing intensive erosion areas or hotspots (S1H), reducing erosion from steep slopes (S1S), and planting strips along stream network in UPA (Strips).

- **Management options targeting gullies (S1G)**

Conservation measures to reduce erosion from gullies, especially in human-accessible landscapes, are recognised as preventive measure for reducing soil loss potential (Tamene et al., 2014; Tamene, 2005). For that purpose, scenario S1G focused on the conservation of gullies and buffer zones of 25 m alongside these gullies (Tamene et al., 2014). Though this study encompasses both natural (protected or not) areas and human dominated landscapes (agrosystems), management efforts through S1G aim at terracing the 25 m buffer zones by converting them into vegetated lands (woodlands), suggesting a status of these areas as protected against human impacts. Therefore, it is proposed as default values for *P* factor (0.5) and C factor (0.01). C factor meets the proposed value for woodlands based on the patterns of the study area where most of the streams and rivers, acting as gullies, are bordered by vegetation. The outputs from this scenario are compared to the benchmark (scenario S0) to highlight the efficiency of the proposed preventive measure.

- **Management options targeting the conservation of erosion hotspots (S1H)**

In this sub-scenario S1H, efforts are undertaken in reducing the erosion severity in the identified erosion hotspots based on acceptable soil loss in the area (Tamene et al., 2014). For the Mo basin with tropical climate, mountainous topography and medium annual rainfall, the tolerable soil loss is located on potential hotspots with soil loss higher than or equal to 15 Mg $ha^{-1}y^{-1}$. The latter threshold was considered to evaluate and easily compare the NSL in the Mo basin with the tolerable limits of 12 – 15 Mg $ha^{-1}y^{-1}$ (Roose, 1996), used in West Africa environments (Le et al., 2012b). In addition, varying threshold for hotspots definition, this scenario evaluated the effects of management size on the NSL at landscape level, if efforts could target soil loss limits lower than (5 and 10 Mg $ha^{-1}y^{-1}$) or higher than (20 and 25 Mg $ha^{-1}y^{-1}$) the acceptable value of 15 Mg $ha^{-1}y^{-1}$. The scenario principle is that erosion hotspots higher than the tolerable set value are assumed to be converted into either vegetated areas (S1HA). In addition, terraces or grasses could be used to conserve gullies along with the enclosure of the erosion-prone areas (S1HB). Proposed default values for P and C factors were 0.5 and 0.01, respectively. The results from this scenario are compared with the benchmark (S0) and other management options to judge the effeciciency of management options on reduction of soil loss.

- **Management targeting exclusively areas with steep slopes (S1S)**

Due to the high roughness of the Mo landscapes, conservation measures focusing on steep slope management are oriented towards the reduction of surface runoff and hydrological processes, which often occur at relatively high rate on these slope positions, regardless of the surface cover. The rate of sediment yield and transport toward rivers/streams is acutely observed in these sensitive lands since slope influences the surface flow rates and sediment movement by increasing surface hydrological phenomena (e.g. Moore et al., 1991). Therefore, efforts targeting these erosion-prone areas are assumed to reduce the amount of net soil loss at the landscape level and reduce river siltation. Though the purpose of this option is not to cut down the steep slopes into gentle ones, it is assumed that preventive measures such as covering these slopes with dense vegetation will stabilise lands and reduce the occurrence of new gullies, and consequently abate the rate of surface runoff and transported sediments (Tamene, 2005). The proposed preventive measures in the current study focused on land with slope higher or equal to 15 ° (considered as very steep). In this scenario (S1S), the considered steep lands (> 15 °) are converted into PA with restricted human interventions affecting their stability. Therefore they are assumed to be covered by relative dense vegetation (e.g. woodlands), setting the C factor value to 0.01. Finally, the outputs from this scenario are compared with other scenario runs and the benchmark to assess the efficient impact of the proposed management option on soil loss/sediment yield at the landscape level.

- Planting strips along river network in unprotected areas

Like terraces and contour lines, strip cropping represents a support practice for soil erosion control. In contrast to structural technologies such as terracing, stonewalls and ridging, strip cropping is part of organic technologies that improve the soil characteristics to resist erosion while increasing biomass production and ground coverage (Donovan & Casey, 1998). A buffer strip of native plants can reduce the impact of surrounding land uses on the sediment yield downstream. This scenario could help in addressing issues of soil loss in undisturbed landscapes that affect biogeochemical cycles of carbon and nitrogen, and implications for climate change issues. In the LAMPT_Mo, it is suggested that the planting of highly diverse native plant species in order to match local soil types and especially increase the resistance to soil erosion (Berendse et al., 2015). Therefore, an alternation of strips and natural vegetation is proposed in the riparian lands up to 500 m alongside rivers (Table 7.3). Exclusively, this option is implemented in unprotected areas to highlight the influence of this support practice on agricultural land use system. In the first 100 m from the riverbanks, the option sets strips of natural vegetation (mainly as riparian forests with heavy-deepen root systems) and the C factor is 0.001. This option is implemented regardless of the stream importance and location in unprotected areas.

Table 7.3. Strip planting in the first 500 m along riversides

Buffer zones	Strip types	C factor and Ri
0 – 100 m	Natural vegetation (Riparian forests)	C=0.001; Ri = 0.35
100 – 200 m and 300 – 500 m	Perennial croplands/orchards /Agroforests	C= 0.15; Ri = 2.13
200 – 300 m	Natural vegetation (woodland/savannah)	C= 0.01 ; Ri = 0.40

7.2.5. Validation of model outputs

In inaccessible mountainous and rural areas data scarcity is a crucial problem when analysing LD. Required data are not available for calibrating and validating preliminary studies of LD. Yet, information on LD, especially soil erosion processes, is necessary for landscape management and rural development. In the case of the Mo basin, this first attempt of modelling soil erosion is tremendously hectic because measurement networks and field experiments on runoff-plots for quantifying the effects of different factors on soil loss are lacking. In addition, there was a lack of experimental data for proper calibration of the model and validation of outputs. To overcome this data paucity and propose the first soil erosion modelling for rural landscapes in Togo with the case of Mo basin, three approaches are used for the validation of the model outputs and performance. First, the validation by construct approach was used to discuss the validity of the entry data, and hence validate the output information. It assumed that in a process-based and spatially distributed model, valid input data yield valid outputs in a modelling approach without “black-box”. The estimates of this study were compared with general data on the soil erosion range over West Africa and other similar mountainous environments of Africa (Symeonakis & Higginbottom, 2015) as well as the tolerable soil loss (Roose, 1996; Le et al., 2012b), assuming that similar environmental constraints yield similar range of soil erosion phenomenon (Tamene & Le, 2015). The historical simulated outputs (1972, 1987, 2000) were compared to the outputs from 2014 set as reference value validated from above (Zhou et al., 2014c). Next, this validation is supplemented by some comparison of the model outputs with data ranges from the surrounding West Africa and sub-Saharan environments (Le et al., 2012b; Roose, 1976; Schmengler, 2010; Tamene et al., 2014). Though some available information from about 50 experimental sites (Roose, 1976) are very old and not specific to the Mo catchment, some recent attempted records exist for West African environments (Hiepe, 2008; Le et al., 2012b; Schmengler, 2010). Meanwhile, soil loss threshold of 10 - 12 Mg $ha^{-1}y^{-1}$ for the tropics (Palmer, 1991) was compared to the model outputs in order to evaluate the soil loss severity in the region. Finally, since the purpose of the soil erosion model is not exclusively the quantification of the amount of NSL but also the capability to provide erosion severity patterns helpful to management options (Le et al., 2012b), the current study also aims at providing a most plausible delineation of erosion severity patterns at the Mo landscape level. Hence, a semi-qualitative approach was used through selective field observations in the study area to judge and match the quality of soil erosion on field with the simulated outputs. The lack of reliable field measured data makes it difficult to assess the extent and severity of erosion in complex landscapes. Stakeholders' perceptions, field scoring and ranking of the erosion factors and evidences of soil erosion can be used for that purpose. Since each natural spatially explicit phenomenon is a result of an association of factors, it is assumed a scoring of such factors or association of factors can help in representing such phenomenon. Therefore, semi-quantitative (field characterization) data were used to assess the capability of the models to predict the amount of soil loss (gross and net). The approach consisted of ranking and scoring individual potential factors of erosion sensitivity

(Appendix 11) during field visits. Prior to the field visit, two hydrological subunits (Tchamou and Boualé catchments) derived using DEM SRTM and drainage networks (Tamene, 2005), were selected based on their accessibility and heterogeneous characteristics. Next, the list of landscape features describing on-site (GSL) and off-site (NSL) was used to characterise each of the selected sites for matching the model outputs and the field observations. Further, the total score calculated for each site was compared the GSL and NSL using regression trends. This approach of field characterisation is widely used to validate soil erosion models (De Vente et al., 2005; Tamene, 2005). Results from the two HRUs could be extrapolated to the entire Mo basin assuming that similar site with similar characteristics undergo similar range of erosion risk potential. The concepts of similar environmental constraints envelops (SECEs) (Tamene and Le, 2015) are concrete explanation of the approach.

7.3. Results and discussion

7.3.1. Historical soil loss in the Mo basin: business-as-usual (BAU) scenario

The estimation of the historical soil loss and sediment yield for the Mo basin was made under the scenario "Business as usual (BAU)" with an emphasis on impacts of LUCC.

a) General trend in the historical NSL in the Mo basin

Figure 7.4 shows the spatial patterns of the GSL and NSL for the years 1972, 1987, 2000 and 2014 in Mo basin. Simulations showed increasing average GSL (about 160, 175, 186 and 279 Mg $ha^{-1}y^{-1}$) for the respective successive years of study. These patterns proportionally aligned with the trends of LUC spatial configuration in the basin (See Chapter 5). The adjustment of the GSL with the sediment delivery ratio (SDR, average values of 5 - 6 %) showed that net soil loss (NSL) patterns are highly influenced by the river/stream network in the catchment (Figure 7.5). Consequently, the average NSL were of 26, 23, 27 and 44 Mg $ha^{-1}y^{-1}$, respectively for 1972, 1987, 2000 and 2014 (Figure 7.4). These results generally shown an increasing trend in net soil loss over time which can be exacerbated by LUCC. For the entire catchment, LUCC inducing vegetation decline exhibited increasing GSL, indicating that an increase soil loss is associated with poor land cover (Figure 7.5) or land cover quality decline as reported by Feng et al., (2010). It is therefore evident that the vegetation cover is important in reducing on-site soil erosion (Feng et al., 2010; Lal, 1993). Similarly, Meshesha et al. (2012) found in Central Rift of Ethiopia that the increasing vegetation degradation resulted in a substantial increase soil loss over time. Furthermore, Berendse et al. (2015) revealed that healthy plant communities and high species richness minimise soil erosion potential on slopes.

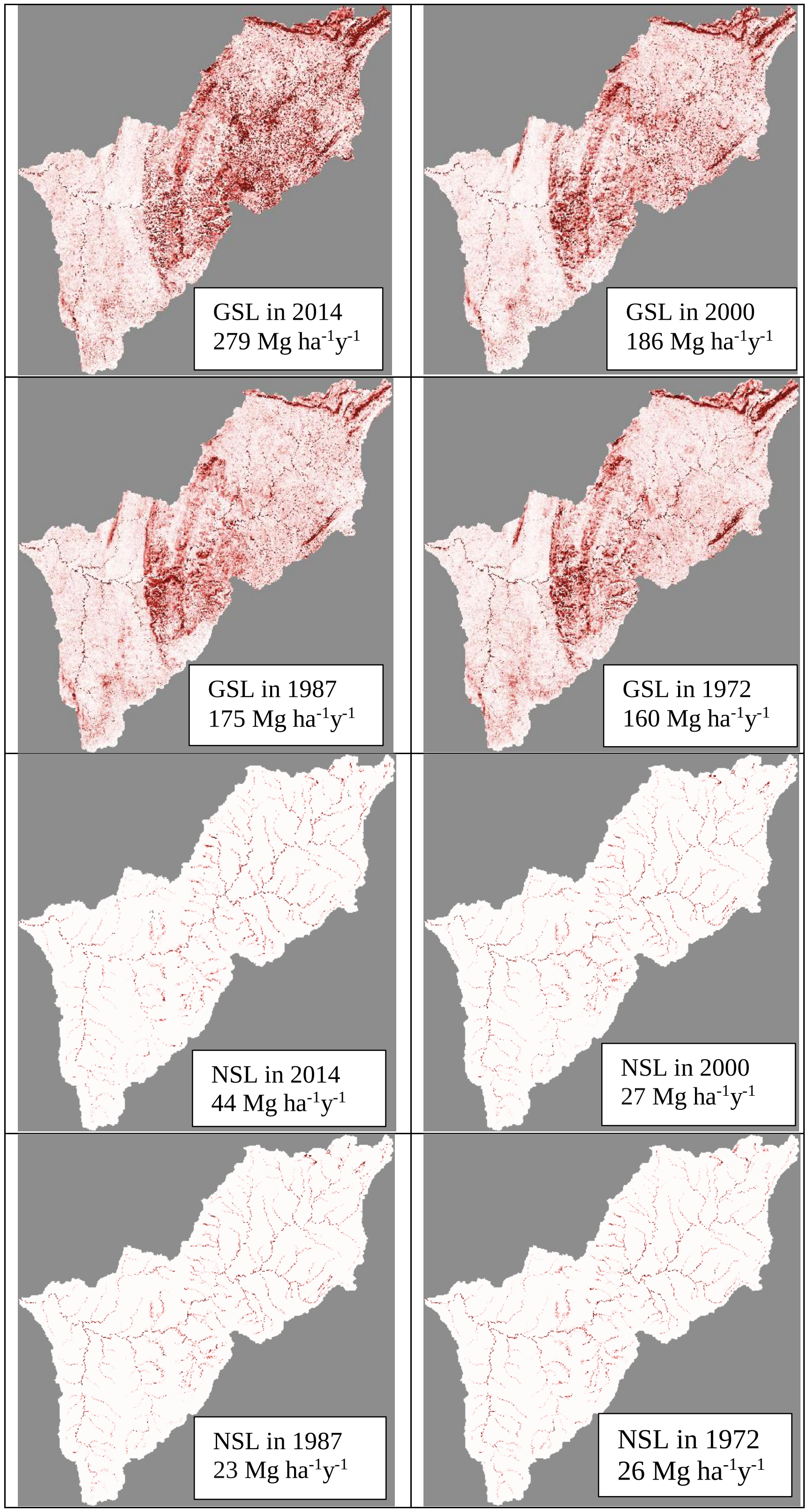


Figure 7.4. Spatial patterns of simulated historical GSL and NSL for the Mo basin

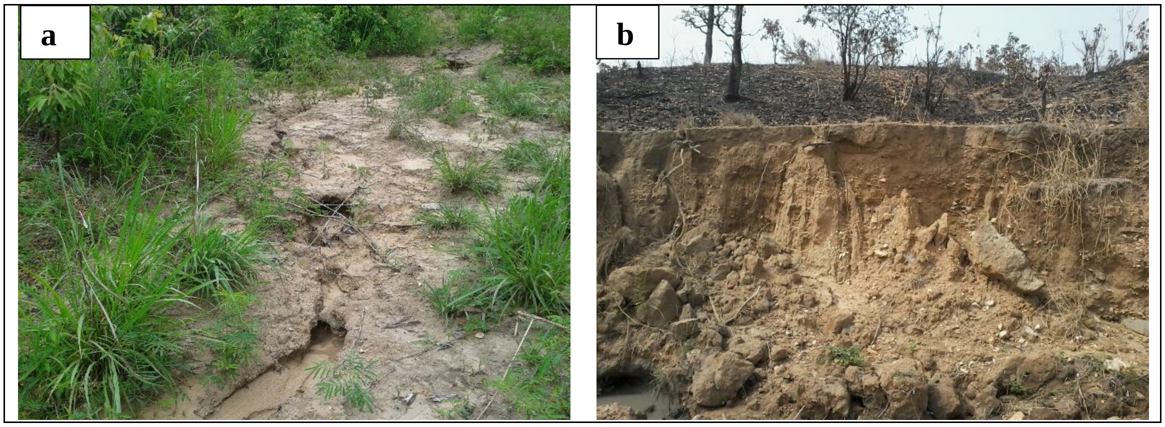


Figure 7.5. Gully initiation (a) and bank collapse (b) on poorly covered lands

The simulated results of this study lie within the soil loss ranges over West Africa (Le et al., 2012b; Tamene and Le, 2015) and similar mountainous environments of Africa (Symeonakis and Higginbottom, 2015; Tamene et al., 2014). The NSL of 2014 is quite high and compared to the modelled value obtained (35 Mg $ha^{-1}y^{-1}$) for White Volta sub-basin in West Africa (Le et al., 2012b; Tamene and Le, 2015). However, field measurements in some sub-catchments of the same White Volta basin showed that the simulated NSL of the current study were quite high and up to the double of the values measured at Doba, Zebila and Bugri (about 19, 27 and 18 Mg $ha^{-1}y^{-1}$, respectively). The relatively high NSL of this study could be due to two intrinsic factors: roughed landform and the sediment routing approach used to adjust the GSL in NSL (Gallant & Wilson, 2000; Tamene et al., 2006). Derived from the Multiple Flow Direction (Freeman, 1991), the flow path length was a function of the stream network heavily developed in the mountainous Mo basin. Furthermore, higher amounts of NSL were expected but the large size of the watershed could have induced a loss of sediment into pits/sinks all over the watershed (Shi et al., 2014). However, in comparison to the tolerable soil loss of 12 – 15 Mg $ha^{-1}y^{-1}$ (Roose, 1996; Le et al., 2012b), current study yielded high NSL (over 25 Mg $ha^{-1}y^{-1}$) though there exist the protected areas. This could be due to the relief of the basin, which causes high sediment yield even in PA. This result indicates the necessity to undertake sustainable practices for reducing soil erosion. Based on the principle of validation by construct, the accuracy level of the model inputs such as LUC types (Chapter 6), and soil erodibility (Le et al., 2012b), and the landform-based inputs (Chapter 4) are satisfactory for modelling (Aguirre-Gutiérrez et al., 2012; Leh et al., 2013; Monserud & Leemans, 1992). Thus, NSL for the Mo basin are quite reliable to guide decision for soil erosion monitoring.

The relatively high NSL of this study could be due to the sediment routing approach using to adjust the GSL in NSL (Gallant & Wilson, 2000). Derived from the Multiple Flow Direction (Freeman, 1991), the flow path length was a function of the stream network heavily developed in the mountainous of Mo basin. Furthermore, it was expected higher amounts of NSL but the large size of the watershed could have induced a loss of sediment into some deposition at intermediate sites' - like low-lying areas and field boundaries - all over the watershed (Shi et al., 2014). However, in comparison to the tolerable soil loss of 12 – 15 Mg $ha^{-1}y^{-1}$ (Roose, 1996; Le et al., 2012b), this study yielded high NSL (over 25 Mg $ha^{-1}y^{-1}$) attributable to mountainous relief with some poor cover and exposed bare surfaces in the basin, especially in savannahs and slopes on top-slope and mid-slope.

Field characterisation shows poor agreement with the predicted NSL and GSL for both subunits and the whole catchments (Table 7.4). However, GSL showed better agreement with the factor scoring approach (FSA) outputs while differences between Boualé and Tchamou indicate the heterogeneous patterns of soil loss in the Mo landscapes. The low agreement may be due to two main reasons. First, there could be biases in the scoring of evidences of soil erosion which could reflect in when compared to the modelled NSL. Second, sediment routing approach that only consider flow path to river, affecting the SDR, induced biases in computing the NSL. Terrain attributes such as soil types, that is specific to each site may also explain differences in the NSL outputs. However, the positive correlations indicate that the model mimics the landscape behaviour despite its heterogeneity. The FSA however indicates that soil erosion is a real phenomenon occurring in the Mo basin with different spatial patterns.

Table 7.4. Correlation coefficients between sum of field scores and predicted soil loss

Hydrological units	GSL	NSL
Mo basin (*n* = 87)	0.135	0.133
Tchamou unit (*n* = 36)	0.225	0.011
Boualé unit (*n* = 51)	0.413	0.344

Note: "n" are the number of field points characterised during factor scoring approach.

b) Distribution of historical NSL in relation to slope classes in the Mo basin

Figure 7.6 shows that steep areas (≥ 15 ^{O}) contributed more to the average annual sediment yields. Areas with lower slopes (< 15 ^{O}) yield NSL lower than 10 Mg $ha^{-1}y^{-1}$ while lands with slope between 5 - 10 ^{O} globally experienced NSL lower than 5 Mg $ha^{-1}y^{-1}$. Higher NSL experienced by flat terrain (< 5 ^{O}) compared to 5 - 10 ^{O} class can be explained by the impacts of human settlements and agricultural fields in UPA and the vulnerability of soils to erosion in PA (see K factor map with higher erodibility in PA; Figure 7.3). Terrain slope gradient in hilly areas has been identified as an important factor that influences soil moisture and surface hydrological processes such as water flow paths (Lv et al., 2013; Penna et al., 2010) and subsequent soil material detachment and transport downstream (Tamene, 2005). Significant positive correlation between terrain slope and sediment yield occur in heavily rugged landscapes (Moore et al., 1991; Zhao et al., 2015). These patterns showed that while flat terrain experience NSL mostly attributable to human influence, large areas of Mo basin are dominated by steep slopes inducing soil loss, which can be exacerbated by improper land use and absence of management interventions.

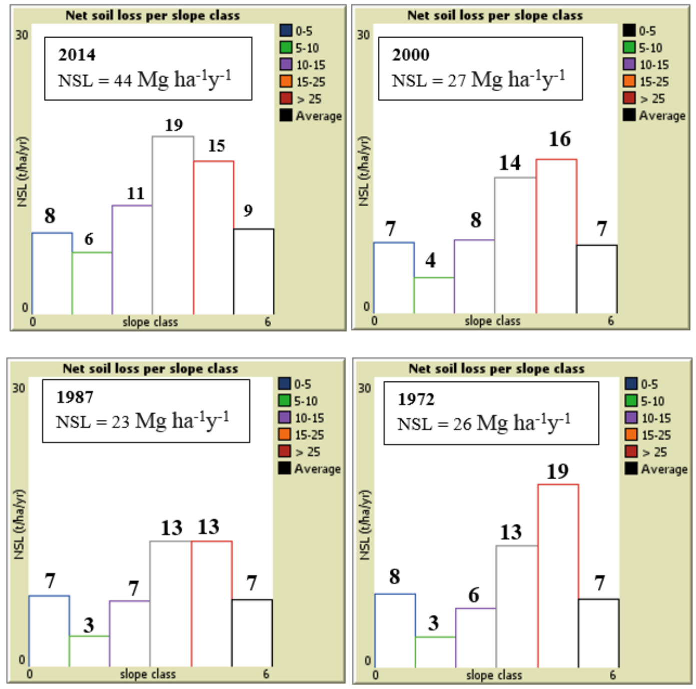


Figure 7.6. Historical NSL per slope classes for the Mo basin

c) Distribution of historical NSL according to LUC types in the Mo basin

The distribution of NSL according to LUC types (Figures 7.7a, b, c & d) showed that savannahs and croplands experience highest NSL over time. There is an increasing trend of NSL with a decreasing canopy cover, ranging from the lowest average values of 2 – 4 Mg $ha^{-1}y^{-1}$ in forests to 16 – 34 Mg $ha^{-1}y^{-1}$ in farmlands. The highest average values of NSL were observed for the most recent years for all cover types except woodlands with an unchanged value of 6 Mg $ha^{-1}y^{-1}$ (Figure 7.7b). Though UPA experience the highest NSL in almost all the cover types, the average NSL in PA for each LUC is quite high with regard to the UPA. The increasing NSL for forest areas over time can be explained by the fact that forest expansion occurred in riparian forests located on erosion-prone streamsides in both PA and UPA. Meanwhile, high NSL in savannahs as natural lands is attributable to not only their low surface cover but also to their topographic locations. Savannahs often occur on exposed top-slopes and mid-slopes with high erosion potential (See Chapter 4). Except forests, all the LUC units in both PA and UPA experience NSL higher than the limit of 5 Mg $ha^{-1}y^{-1}$, probably due to their high cover. However, according to reported NSL limits of 12 – 15 Mg $ha^{-1}y^{-1}$ (Roose, 1996) used in West African environments (Le et al., 2012b), the high NSL areas were specifically croplands/ bare soils, and to some extent savannahs. Overall, this stratification of NSL identifies the most contributing LUC types to sediment yield in the catchment, calling for efforts to reduce erosion potential in cultivated lands and top-slopes as well as riverbanks in UPA.

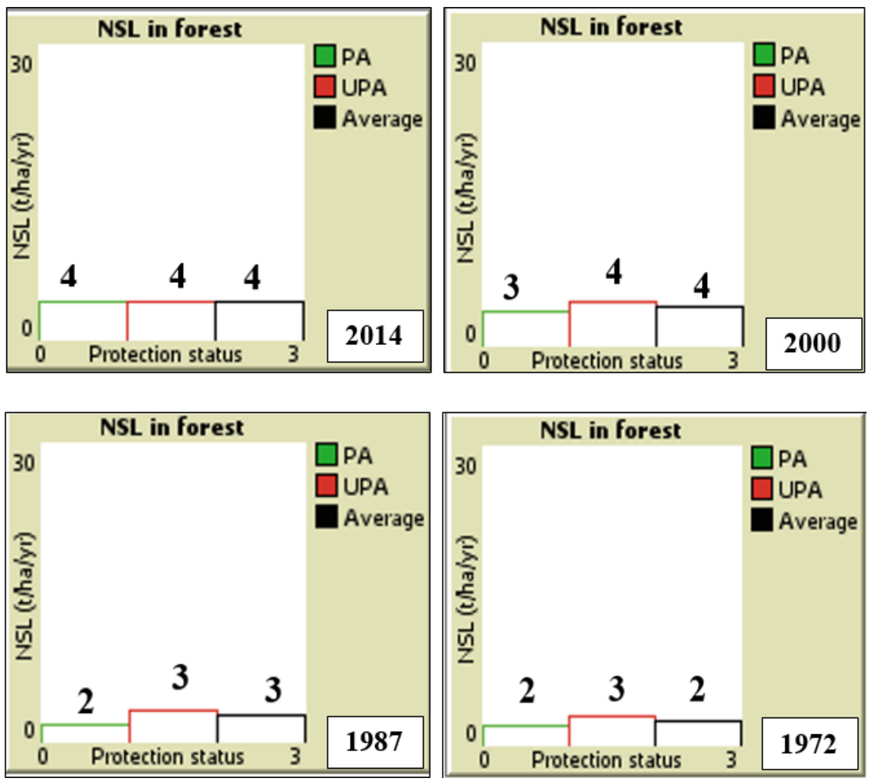


Figure 7.7a. Historical NSL in forests according to land protection status

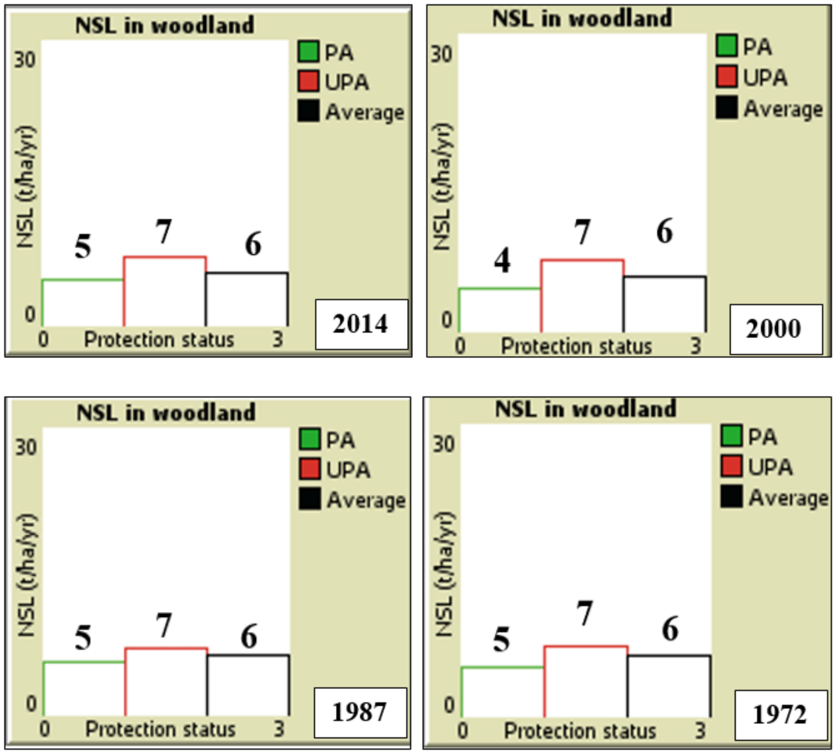


Figure 7.7b. Historical NSL in woodlands according to land protection status

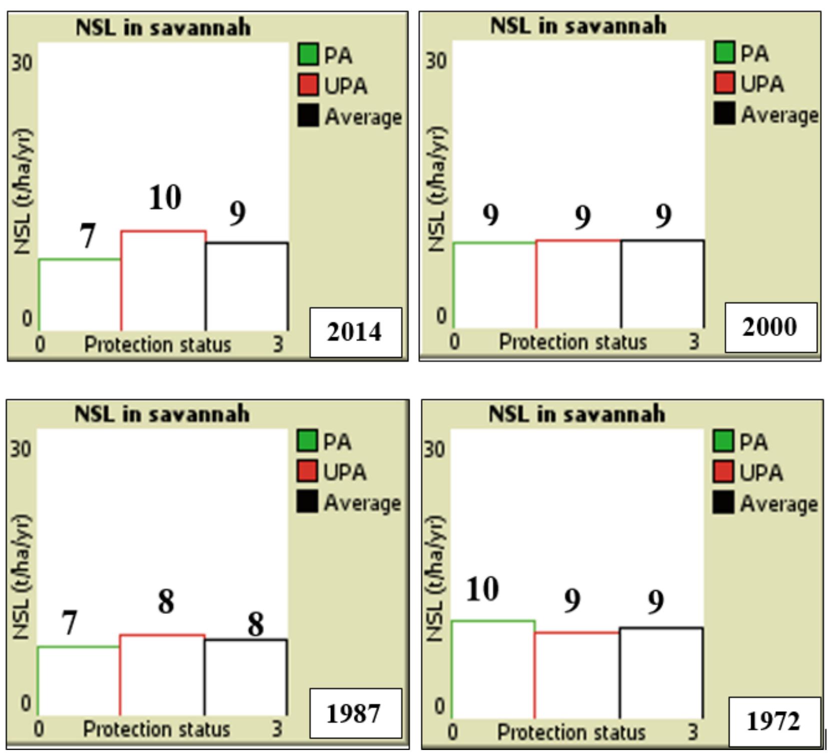


Figure 7.7c. Historical NSL in savannahs according to land protection status

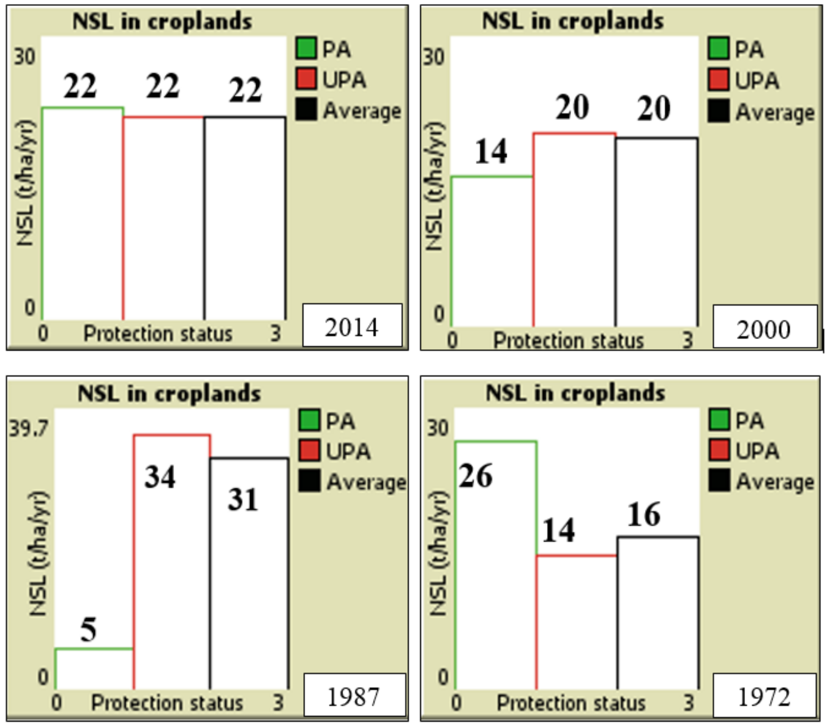


Figure 7.7d. Historical NSL in croplands according to land protection status

d) Historical NSL in relation to distance to river in the Mo basin

Figure 7.8 shows the historical distribution of mean NSL estimates in relation to buffer zones along river network. With exception to the buffer 0 - 50 m, the proximity analyses show a historical increasing trend in NSL for all distance classes, though the range of the average NSL (7 - 9 Mg ha^{-1}y^{-1}) did not show much change. The closest 50 m to rivers/streams yield very large NSL (79, 76, 70 and 66 Mg ha^{-1}y^{-1}, respectively for 1972, 1987, 2000 and 2014), while the farther distances to river experience the sharp lower average NSL. However, the NSL reduction in the first 50 m was concomitant with increasing NSL in other distances up to 200 m set for observation in the current study. In all cases, only the first 50 m, which lay within the effective riparian zones in savannah-dominated landscapes, experience sediment yield beyond the reported limits of 12 - 15 Mg ha^{-1}y^{-1}. This observation can be explained by the high potential of erosion on riverbanks, especially in unmanaged landscapes. In addition, increasing NSL in other buffer classes indicate the expansion of erosion severity for lands located in further from rivers/streams, especially the adjacent 50 – 100 m and 100 – 150 m) that reach the levels of 12 - 17 Mg $ha^{-1}y^{-1}$ in 2014. At the closer positions to river, soil erosion is more acute, due to bank erosion and heavy gully network in relation with landscape slope (Tamene, 2005) since great proportion of the NSL is from channel banks (0 - 50 m; Figure 7.7) with high probability of sediment delivery to channels. This proximity analysis demonstrates that areas located up to 100 m alongside streams are potential targets to concentrate management efforts for controlling soil erosion in the basin.

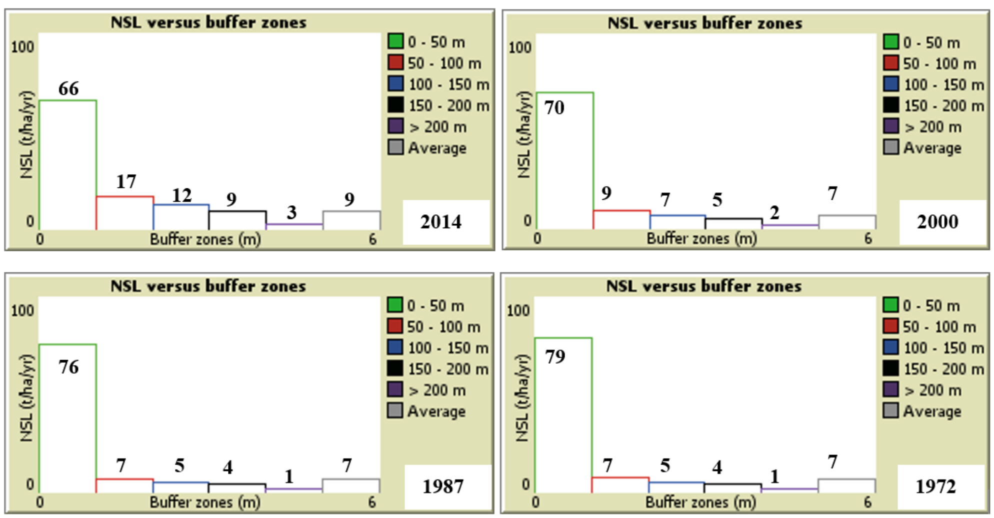


Figure 7.8. NSL over time according to distances to river/stream

7.3.2. Options for erosion control in the Mo basin: land management scenarios

a) Effects of management options on NSL in relation to slopes

Figure 7.9 shows the graphical outputs of the effects of different land management options on the distribution of NSL in relation to slope classes. In reference with the baseline, the scenarios targeting the gullies (42 Mg $ha^{-1}y^{-1}$) and the development of strip (41 Mg $ha^{-1}y^{-1}$) were less efficient for the NSL reduction in the landscape (5 and 7 %, respectively). This is probably because steep slopes are quite stable over the landscape, which is more under protected status. NSL is marginally sensitive when strips of 100 m are planted interlined with natural stands of 100 m, indicating the inefficiency of this management option. Meanwhile, three options were efficient: S1S, S1HA and S1HB. Enclosing exclusively erosion hotspots (S1HA) and combining hotspot enclosure with terraces (S1HB) induce significant reduction in NSL to about 15 and 13 Mg $ha^{-1}y^{-1}$, respectively. The efficiency was about 66 and 77 % reduction for the S1HA and S1HB, respectively. However, the quite similar effects of S1HA and S1HB is due to the fact that less erosion hotspots are located within gullied areas. Outputs from S1S indicated that steep slopes ($\geq 15^{O}$) contributed about 64 % to the total NSL in the Mo basin (reduction from 44 to 16 Mg $ha^{-1}y^{-1}$). Large proportions of erosion hotspots are located on slopes $\geq 5^{O}$ all over the landscape (Figure 7.9). The management options (S1HA and S1HB) reduce the sediment yield at proportions ranging from 83 % to 95 % for slope classes of 5 – 10 ^{O} and 15 – 25 ^{O}, respectively. Since most of the steep slopes is located within PA, it is assumed that the NSL in those areas are more nature-controlled processes. In contrast, flat areas outside PA located on lower slopes contribute to NSL due to human interferences such as rural trails, which increase NSL if any measure is undertaken (Figure 7.10).

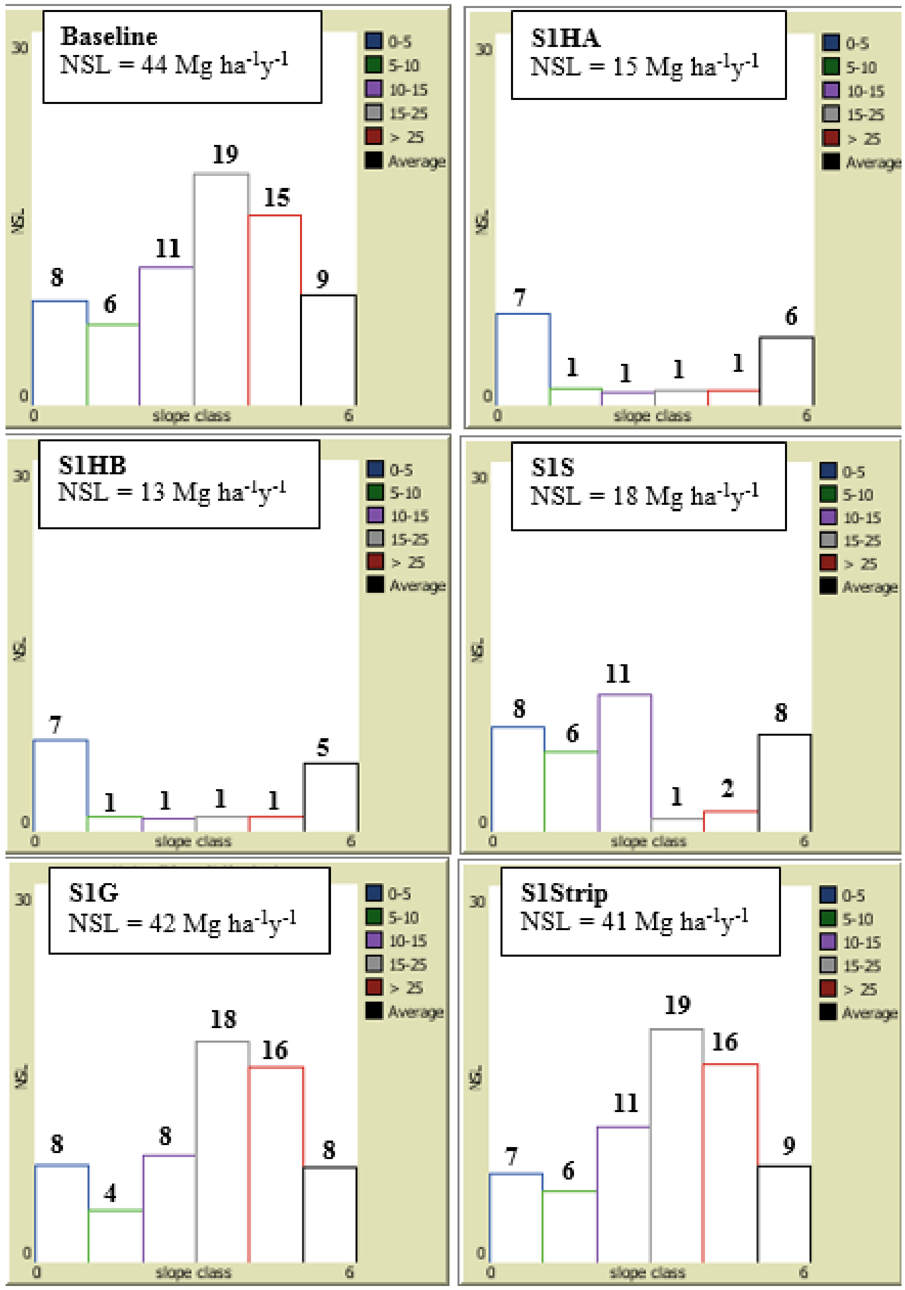


Figure 7.9. NSL for different land management options and the baseline conditions

b) Effects of management options on NSL in relation to buffer zones

According to the buffer zones along riversides, the options S1HA and S1HB targeting erosion hotspots indicated a significant reduction of NSL in 50 m river buffer zones (Figure 7.10). However, the soil loss within the 0 – 50 m buffer remain quite high due to the fact that these areas are more sensitive to erosion, especially lateral erosion. Compared to the baseline, management options promoting strip planting (Strip) and enclosure of steep slopes (S1S) did not significantly reduce sediment yield in any of the buffer zones. Overall, S1S and Strip reduce marginally the average NSL up to 5 and 7 %, respectively. Meanwhile, the reduction ranges from 63 % to 70 % when S1G, S1HA, and S1HB are implemented. The 25-meters buffers often correspond to riparian lands, which experience high bank erosion (Figure 7.11) with consequent river enlargement (Douglas & Guyot, 2005). Yet the role of enhancing vegetation cover and species richness is regarded important in reducing the potential of soil loss (Berendse et al., 2015).

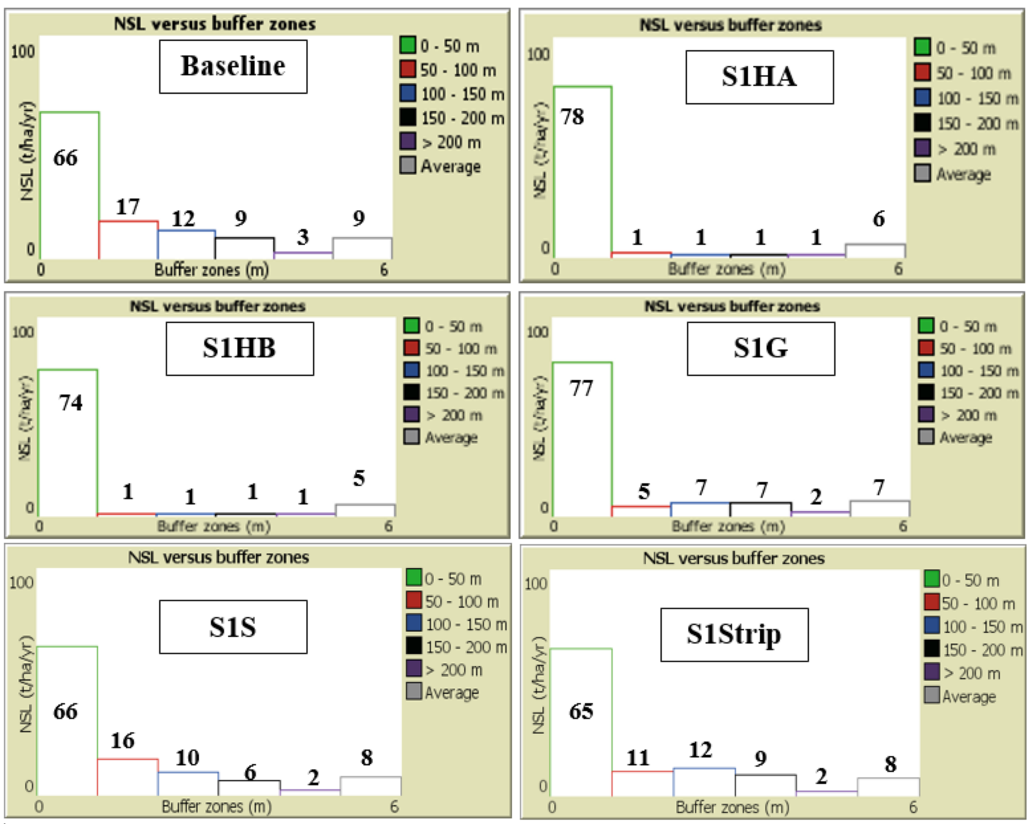


Figure 7.10. NSL according to river buffer zones for different management options

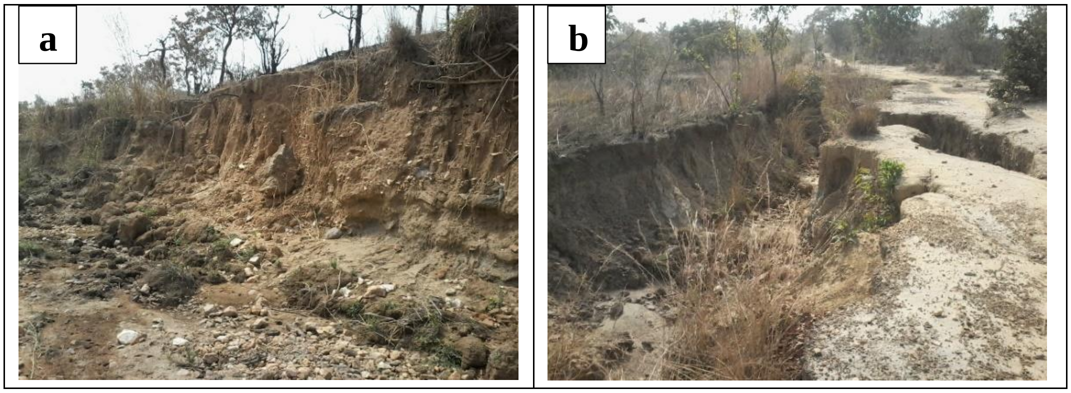


Figure 7.11. Collapsing riverbank in Tchamou River (a) and gullies initiated along a rural trail on moderate slope in Tchamou catchment (b)

c) Effects of management options on NSL in relation to LUC types

The reduction of NSL in forest areas is almost 100 % for the options S1HA, S1HB, and S1G (Figure 7.12). Average NSL in forested areas declines from 4 Mg $ha^{-1}y^{-1}$ to almost 0 Mg $ha^{-1}y^{-1}$ (not absolute 0). Meanwhile, compared to the mean NSL of the baseline option, the average NSL for S1S and Strip options do not change in term of erosion severity in forest areas at the landscape level. The latter options do not induce any change because forests merely occur on steep slopes in the Mo basin; rather they occur at riverbanks and inland valleys (See Chapter 4). This is also evident through strip planting which does not affect the first 100 m buffer where most of the forests occur.

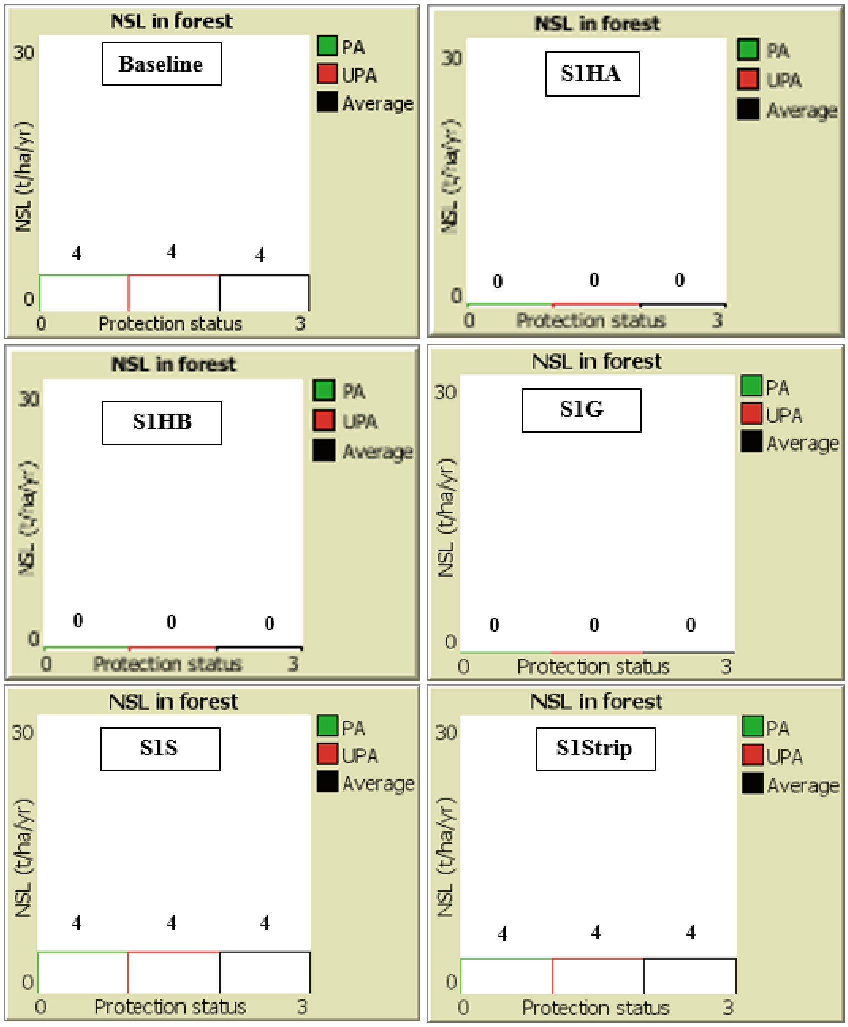


Figure 7.12. NSL in forests according to protection status and management options

Figure 7.13 presents the NSL distribution in woodlands for the different land management options and land protection status. Options S1HA, S1HB and S1G induce an increase of soil loss under woodlands. The increase is highly significant with the option S1HA, suggesting that the option tend to decrease NSL in other LUC units while increasing the soil loss in woodlands. This is because S1HA is an option that converts all erosion hotspots into similar patterns prevailing in the woodlands of the baseline (See Section 7.2.2.2). UPA of all options exhibit the highest NSL, while planting strips and S1S show quite stable NSL in line with the baseline. Except S1S, none of the options significantly reduce the NSL in woodlands of PA and UPA.

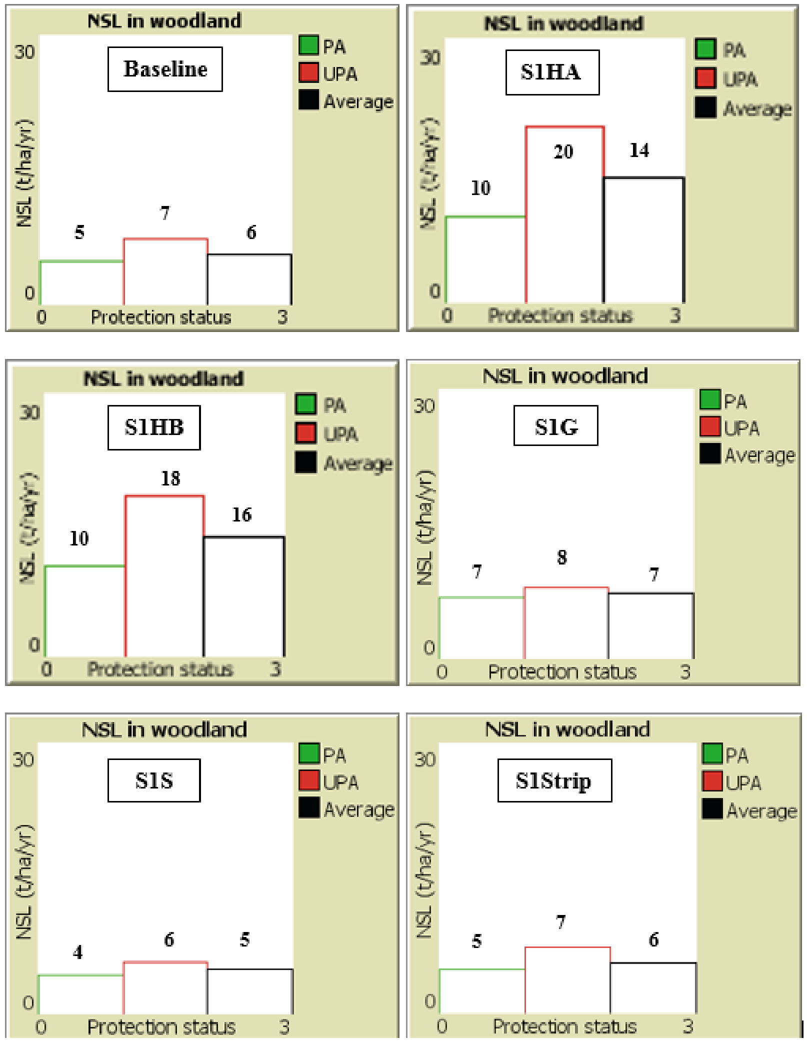


Figure 7.13. NSL in woodlands for different management options and protection status

The impacts of the management options on NSL in savannah are shown in Figure 7.14. Options S1HA and S1HB induce similar decreasing effects on NSL in savannahs for both PA and UPA. Up to 90 % of the baseline NSL is reduced with the implementation of the erosion enclosure with optional terraces. Though S1G and S1S show a slight decrease of savannah-NSL in both PA and UPA, the effects are very low to encourage the adoption of such options towards the reduction of savannah-specific NSL reduction.

Savannahs/shrubs often occurs on top-slopes (See Chapter 4) with high erosion potential, indicating the significant effects of hotspot enclosure on NSL. Consequently, planting strips do not affect in any case the NSL of savannah since the strip bands concern only a distance greater than 200 m from both side of the river/stream network.

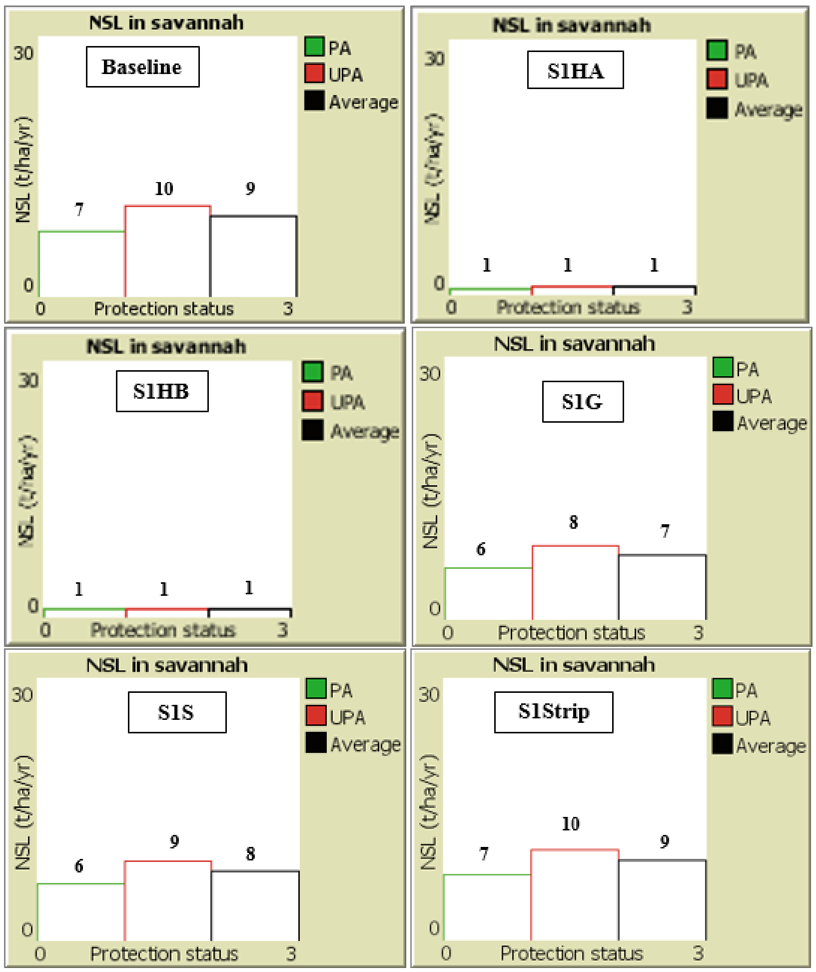


Figure 7.14. NSL in savannahs for different management options and protection status

In areas with poor surface cover dominated by croplands, S1HA and S1HB could significantly reduce the amounts of the average NSL from 22 to 1 Mg ha^{-1}y^{-1} (Figure 7.15). The reduction is more significant in UPA than PA, because croplands in UPA are more located on flat terrain where the NSL threshold (15 Mg ha^{-1}y^{-1}) is more or less controllable.

Strip planting in UPA contribute to a significant reduction of NSL (from 22 to 9 Mg $ha^{-1}y^{-1}$), indicating that this option is effective in reducing soil loss up to 59 % compared to the baseline. The NSL in UPA remains unchanged for croplands because the strip-planting option suggests that sole areas outside are managed since croplands in PA are illegal incursions that will not gain agreement for option implementation. Option S1S seems to be not efficient in reducing NSL since it is rare to observe croplands on steep lands. When land management aims at targeting gullies, NSL in PA significantly reduces by 50 % whereas in UPA, the change is not sensitive at landscape level. This is because gullies are densely developed in PA, which lies in more rough landscapes while in UPA, farmers concentrate on more fertile lands on flat terrains and inland valleys.

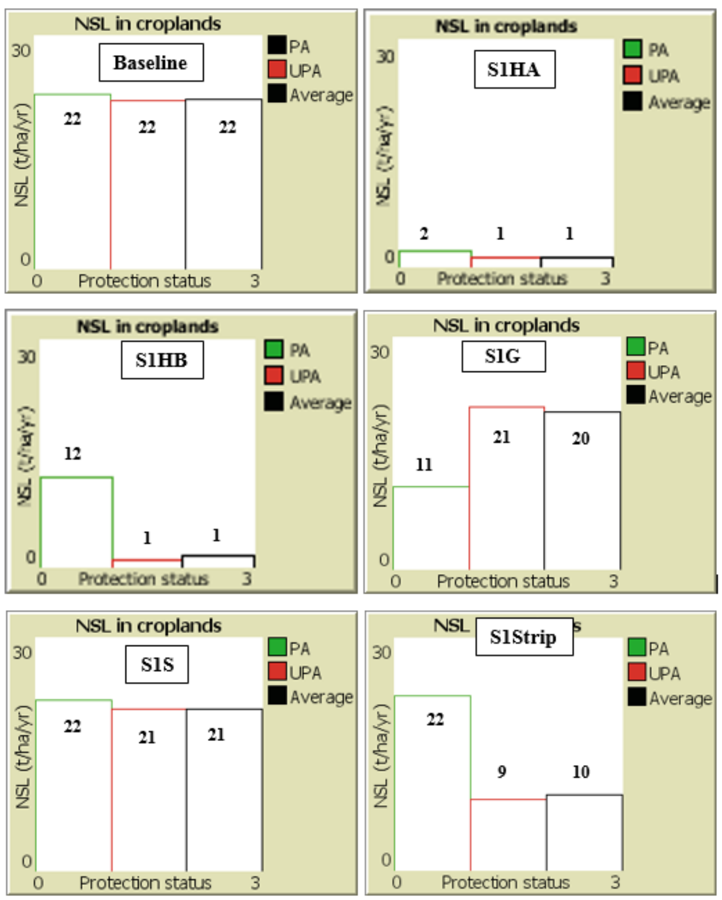


Figure 7.15. NSL in croplands for different management options and protection status

Though all LUC types are erosion-prone, soil erosion is manifest when land surface cover decreases through land uses (e.g. agricultural lands). In this context, Labriere et al. (2015) suggested that certain types of land uses should be avoided and sound practices of soil and vegetation management (e.g. contour planting, vegetative buffer strips) be implemented to reduce soil erosion up to 99 %. Interventions for erosion reduction in the Mo basin will be more effective when targeting hotspot areas, mostly located on gullied lands. This study showed an efficiency of management options of up to 66 % and 59 %, respectively when erosion hotspots and gullies are terraced and grassed. Depending on the soil loss threshold defined for the management purpose, the consequent NSL tend to decrease when the threshold increases (Figure 7.16), especially for 2000 and 2014. For these latter years, consequent NSL was lower after combination with erosion hotspots though the NSL values varied between 14.94 and 15.80 Mg $ha^{-1}y^{-1}$ for all years. This can be explained by the fact that LUC of those years were of lower coverages with high NSL that were very sensitive to the management while in 1972 and 1987, the soil coverage was quite efficient that the proposed management do not improve the erosion control. Hence, it is concluded that the proposed management options are efficiently perceptible for highly human-degraded or -transformed landscapes.

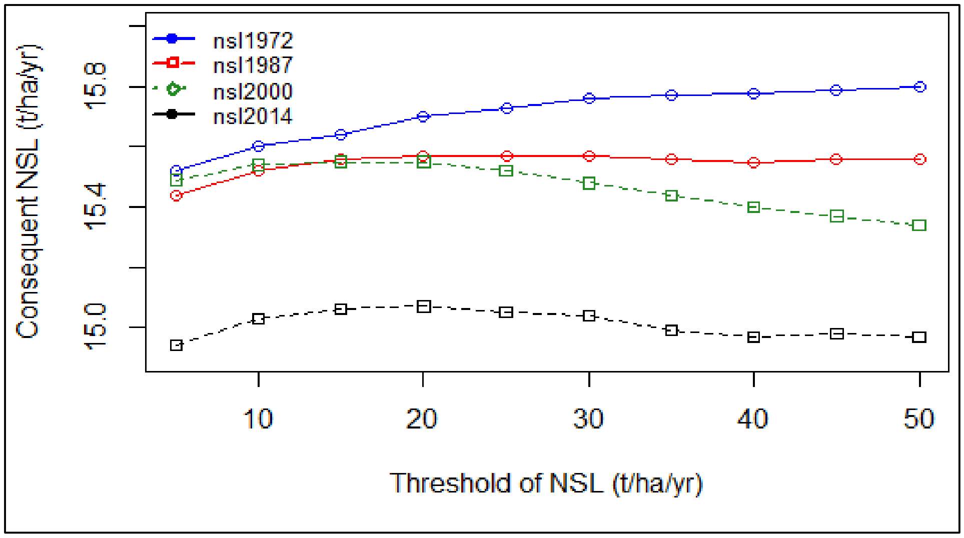


Figure 7.16. Consequent NSL in relation to soil loss thresholds for the four years

7.3.3. Limitations of LAMPT_Mo and contributions to adapted land management

The LAMPT_Mo tool allowed the simulation and quantification of soil erosion for Mo basin. It involved the assessment of the relative contribution of each LUC type, the proximity to river network, the topography and the land protection regime on the average soil loss at the catchment level. This study is probably the first attempt mimicking hydrological processes using such a model in a complex and heterogeneous landscape of the Mo basin. Though model calibration and validation referred to data range from West African environments, it suggested that specific and historical field measurements could have induced more positive impacts on soil erosion measurement for the Mo basin (Tanyas et al., 2015). The model performance of the sediment yield was within the range of values reported in similar studies in West African environments and humid tropics with similar environmental conditions (Tamene and Le, 2015; Tamene, 2005; Le et al., 2012b; Roose, 1996). The necessity of direct measurements of soil loss is of great interest for real phenomenon study but the paucity of such data is often a constraint compelling the use of models to represent the influence of soil and vegetation management on soil loss in humid tropics (Labriere et al., 2015) and poorly accessible regions (Tamene and Le, 2015). It is therefore suggested that this initial study should be supplemented by long term field observations to really capture the behaviour of soil erosion processes at the landscape level, taking into account all landform and land use units as well as the sediment fate measurements at the basin outlet or dams (e.g. Tamene, 2005; Hiepe, 2008; Schmengler, 2010). Furthermore, perspectives of erosion modelling for Mo basin should rely on specific input data such as C factor and R_i coefficients in order to avoid the over- or underestimation of the simulated soil loss (Tanyas et al., 2015; Yang, 2014).

7.4. Conclusion

In this study, processes of soil loss and amount of sediments were simulated under different factors i.e. possible management units (slope classes, LUC types, distance to river, land protection regime) and comparison was made between the different outputs to highlight the trend of soil loss at landscape level and evaluate the efficiency of erosion control interventions. The LAMPT_Mo, which is a RUSLE-based tool, was calibrated for the Mo catchment using common reference data from West African environments whereas the validation of the outputs reasoned based on the output ranges and the tolerable limits required for management interventions. Soil erosion estimates from 1972 to 2014 increased following the spatial-temporal patterns of LUCC and landform. The average annual sediment yield for the Mo basin were of about 26, 23, 27 and 44 Mg $ha^{-1}y^{-1}$, respectively for 1972, 1987, 2000 and 2014. The highest soil loss areas were located on steep slopes ($\geq$ 15 ^{O}), under areas with low vegetation canopy such as savannah and croplands, and in areas closer to riverbeds (distances $\leq$ 100 m). These estimates were quite different from the values obtained by several studies in West African environments. These differences could be caused by the different methodological approaches, the mountainous environment of the Mo basin and possible diversity in land management practices. In this study, various management options showed an efficiency up to 77 % and 66 %, especially when erosion hotspots are enclosed and with gullies terraced grassed, respectively. Enclosures of erosion hotspots have the potential to reduce soil loss up to 90 % for slope classes of 5 – 10 ^{O} and 15 – 25 ^{O}. Meanwhile efforts to enclose steep slopes in the landscapes may reduce NSL up to 64 %. The application of this erosion model in the Mo basin showed sufficient insights in identifying soil erosion-prone areas and judging the severity of the average soil loss in comparison with tolerable limits. For conservation purposes, this study is helpful in addressing soil loss in relation with LUC units, slope classes and distance to river. In perspective, proper calibration and validation of the LAMPT_Mo have to be performed based on field measurements to improve the reliability of the soil loss simulation. Therefore, it is suggested that different modelling approaches should be performed in further erosion simulation in order to allow better comparison and ensure the reliability of the outputs from current LAMPT_Mo.

CHAPTER 8: CONCLUSION AND RECOMMENDATIONS

This chapter summarises the key findings of the study in relation with each specific objective. It highlights the major limitations of the study. Recommendations for policy and further researches are also highlighted in this chapter.

8.1. Summary of research findings

<u>Research objective 1: To determine the contemporary soil conditions in relation with biophysical and human factors in the landscapes of the Mo River basin</u>

During this study, analyses of soil samples collected at the depths of 0 – 10 cm and 10 – 30 cm revealed that SOC ranged from 2.04 % to 3.22 % and 1.78 to 2.23 % in the topsoil and subsoil, respectively. Meanwhile, TN content varies between 0.06 % to 0.16 % and 0.05 % to 0.09 % for the topsoil and subsoil, respectively. These results revealed that SOC and TN were mostly concentrated in natural vegetation, especially forests and woodlands that occur at riverbanks, inland valleys and low-slopes. Meanwhile, agricultural lands (farms and fallows) exhibited low TN and SOC. Topsoil contributes more to TN and SOC for the overall 30 cm depth and their concentration diminishes with increasing soil depth. In relation to landscape positions, the study highlighted that inland valleys, riverbanks, and foot of hills store more SOC and TN over the basin. These patterns were similar for both LUC types and landscape positions, regardless to the land protection regime. Using correspondence analyses, the research revealed that human disturbances negatively correlated with land health and land protection, which appeared as controlling factors having positive effects on SOC and TN. The most important human disturbances affecting the spatial distribution of SOC and TN were bushfire, cattle grazing, and farming. In agrosystems, similar nutrient contents were observed among fallows and farmlands, though the latter are slightly nutrient richer for cultivation. In comparison with the natural and undisturbed lands, the results indicated that cultivated lands still have high amount of nutrients, especially SOC, indicating their current cultivation. In sum, Mo basin has a great potential in SOC and TN storage controlled by various human and landform factors. This important potential could contribute to regulate biogeochemical cycles if efficiently managed, to the benefits of the food security and climate change mitigation.

Research objective 2: To determine the contemporary vegetation patterns in relation with biophysical and human factors in the Mo river basin

Different plant communities occurring in the Mo basin are controlled by a combination of *in-situ* features (both ecological and geomorphic), land protection status and human disturbances at different levels. Each site has its singular attributes, whose interaction with anthropogenous and natural conditions could present similarities/dissimilarities in the landscape patterns. Results from detrended canonical analyses (DCA) showed that protected areas have closer vegetation patterns (three vegetation groups) compared to unprotected lands, which have four groups, probably induced by high human interferences. The structure and diversity of the landscape patterns can be attributed to the differences in these aforementioned environmental and human attributes. High cover vegetation stands (riparian and dry forests) occurred mostly along riversides, inland valleys and flat terrain while poorly covered stands (shrubs and savannahs) occurred mostly on top-slopes. The analyses of the attributes of each vegetation group (stand characteristics, soil condition, and human disturbances) enabled the understanding of the differences in the landscape condition. There was an agreement between good soil conditions (moisture and nutrient richness) and the stand structure and diversity (high canopy coverage and structural characteristics such as basal areas, tree density, diameters, and height). Analyses of human disturbance footprints indicated that the protected areas of the basin are experiencing degradation. Generally, healthy vegetation is located either in PA or in inaccessible UPA such as roughed mountainous areas or far from settlements. The study showed that biological conservation could also target unprotected landscapes since some wild landscapes located in inaccessible and low populated areas still have great potential for conservation. These parameters should be considered in developing landscape management and planning strategies towards landscape conservation in the basin and in Togo as well as the contribution to climate change mitigation.

Research objective 3: To assess the LUC patterns in the Mo River basin over 1972-2014

The third research objective focused on assessing the long-term LUCC important for LD assessment and monitoring strategies. The use of time-series Landsat images for the period 1972, 1987, 2000 and 2014 was helpful to reconstitute the historical landscape patterns in the basin. Supervised classification of these images into six LUC types yielded good mapping results (overall accuracy ranged between 69 and 92 %). Natural lands dominated the whole basin, except the east-northern parts where agricultural and human settlements expand over time. The assessment showed that natural vegetation decreased from 99 % in 1972 to 91 % in 2014. This is quite low due to the important share of the PA in the rural landscapes of the Mo basin. However, the decline could be attributed to agricultural expansion, which areas passed from 0.2 % in 1972 to 8 % in 2014. These changes can be closely linked to the increasing need of the population vis-a-vis land resources. Despite this dominance of natural vegetation, woodlands showed acute areal loss while forests and savannahs substantially increased in their coverage. With increase in land demand for food, and subsequent fuelwood energy, further decline of land cover and quality are expected from these pressures, hence endangering the conservation of the ecosystems and their capacity to provide ESS. The results of this study provide therefore important directions for quantifying these impacts, including soil erosion and soil nutrient dynamic. For reversing the loss of natural vegetation and related ESS, this study provided information that may be important in guiding managers and policy makers in formulating new strategies for integrated land management.

Research objective 4: To assess the impacts of historical LUCC on the contemporary landscape services in the Mo River basin

The quantitative assessment of historical LUCC trajectories provided an understanding of the processes inducing landscape dynamics in the Mo basin. The dominant conversion trajectories are markedly the agricultural land conversion and the degradation of the natural vegetation. The rates of natural vegetation decline in PA were higher during the period 1972-1987 (1.2 %) and 2000-2014 (2.1 %). For all periods, transitions to natural vegetation occurred mostly in the southwestern parts under protection status. Meanwhile, permanent agricultural lands were more observed in the centre to eastern parts of the free access lands, especially in areas around settlements and along road network. With such spatial temporal trends, there is high probability of conversion of more natural vegetation into croplands and poorly covered vegetation (savannah/shrubs) in the future. The assessment of the impacts of such LUCC on SOC and TN showed evidence of nutrient loss in soils continuously cultivated and those undergoing continuous quality declines. It is highlighted that land conversion processes affected mostly topsoil SOC and TN.

However, LUCC trajectories had marginal effects on soil loss patterns, which are rather controlled by landforms. This information could help in understanding the effects of landscape dynamics on contemporary soil ESS and hence, help in designing tools to support adapted land use.

Research objective 5: To model soil erosion patterns and land management options for LD mitigation and landscape restoration in the Mo River basin

The simulation of soil erosion at the Mo basin scale revealed that historical soil loss vary highly in relation with LUC types, landscape position and land management regime. The LAMPT_Mo was calibrated for the Mo catchment using common reference data from West African environments. Despite the challenges related with the calibration and validation, the model was quite capable of reproducing soil erosion patterns, which are fairly in line with the soil loss estimated using field characterisation and estimates from various similar environments in West Africa basins and tropical watersheds. Historical soil erosion estimates for the Mo basin were about 26, 23, 27 and 44 Mg $ha^{-1}y^{-1}$, respectively for 1972, 1987, 2000 and 2014. These estimated NSL were quite higher than reference values and reported soil loss rates in various catchments. It is revealed that steeper slopes, poorly covered lands and closer areas to river network yield high sediment amounts landscape wide. The highest soil loss areas were located on steep slopes ($\geq 15\,^{o}$), under areas with low vegetation canopy such as savannah and croplands, and in areas closer to riverbeds (distances $\leq$ 100 m). This study showed an efficiency of management options up to 64 %, 66 % and 70 %, for managing steep slopes, managing erosion hostpots without and with grasses/terraces systems. Depending on the soil loss threshold defined for the management purpose, the consequent NSL tend to decrease when the threshold increases. Enclosures of erosion hotspots have the potential to reduce soil loss up to 90 % for slope classes of 5 – 10 ^{O} and 15 – 25 ^{O}. Meanwhile efforts to enclose gullies in the landscapes may result in a reduction of soil loss up to 5 %, indicating most of the erosion hotspots in the Mo basin are located outside gullies. The application of this erosion model in the Mo basin showed sufficient insights for judging the severity of the average soil loss in comparison with tolerable limits. It is concluded that LAMPT_Mo is a spatially distributed modelling tool capable of providing an insight into the erosion/deposition patterns, implementing and identifying best management options in supporting land conservation decision-making.

8.2. Limitations of the research

Despite the interesting outputs, some limitations arose, especially from the multidimensional approach used in the study:

First, the sampling approach and the data used in this study could have been more extensive by drawing higher and equal or proportional number of plots in PA and UPA. In addition, instead of using solely data from natural and geoinformation sciences, integration of social dimension could offer better appraisal of the landscape change.

Next, the study suffered of separating natural processes of landscape change from human-induced transformation over time and space in the basin. This could have offered real understanding of the role of human in the landscape change in the Mo river basin.

Third, since some of the indicators were based on observation and estimation, the individual subjectivity can be a limitation to this study. For instance, expert judgement used in the scoring of the factors of soil erosion could be sources of biases in analysing and validating LD assessment.

Fourth, The land use decision making in the study area was less captured as the implemented scenarios of land management in the current LAMPT_Mo is purely based on the assumption that stakeholders could find appropriate the proposed options. Information from key informants and household surveys could have provided more conclusive recommendations regarding the likelihood of such scenarios to be realisitc.

Finally, the validation of the LAMPT_Mo is still a major challenge not yet addressed rigorously in this study. Though the study compared the model outputs to the observed regional data, comparison with observed data for the particular Mo basin was a major limitation beyond the scope of this study.

8.3. Recommendations

8.3.1. Recommendations for policy

From the study, the following recommendations are formulated to support policy and decision-making for sustainable land management in the Mo basin.

1) Landscape approach has shown considerable potential in detecting LD and fragmentation patterns in both PA and UPA. Therefore, it is suggested law reinforcement in PA and the development of good agro-sylvo-pastoral schemes outside PA in order to promote land conservation, to meet social and environmental objectives at local and national scales as well as ongoing global challenges.

2) The present study demonstrated that the spatially explicit approach that combined field surveys and legacy information could improve the understanding of landscape dynamics and identification of degradation-prone areas. This approach can be applied with other landscapes towards the promotion of cost-effective restoration and conservation efforts.

3) The use of legacy data in combination with contemporary information coupled with modelling is invaluable step for monitoring LD and supporting decision-making. However, challenges often emerge in get legacy those data on specific landscape conditions. Efforts aiming at developing database on the landscape behaviour (land information) would therefore be of great interest to support science for development.

8.3.2. Recommendations for further research

Certain fundamental questions are still unanswered through this study carried out in the multifunctional landscapes of Mo river basin.

1) LD is a multidimensional problem emerging from a combination of multiple actors/factors. It is therefore desirable to expand this research on the perception of local stakeholders in order to offer a better appraisal of the rural environmental changes and the causes in relation to local adaptation and mitigation strategies.

2) Landscape dynamics influences the provision of ESS (i.e. carbon storage and sequestration, surface water quality and regime, and landscape resilience), and their future behaviour. Further investigation could use permanent measurement plots, and newly available free satellite images of better resolution (ASTER, Sentinel) to offer a better monitoring of the landscape dynamics in both PA and UPA.

3) Exploratory analyses provided historical and contemporary views of soil erosion patterns in relation with LUCC in the Mo River basin. Interest should be given to scenarios of management that consider future trends of LUCC and climate change in order to proactively propose more integrative alternatives to mitigate trade-offs and improve synergies between the landscape dynamics and the provision of ESS.

4) It will be interesting to expand the current LAMPT_Mo into a multi-agent system (MAS) that to couple and link the human and environment systems (i.e. land use decision-making and management and soil-landscape system) and in order to holistically represent the landscape dynamic in the Mo river basin.

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APPENDICES

Appendix 1. Canonical and correlation coefficients of the first four axes of DCA of the 75 relevés (Figure 4.2)

Variables	Canonical coefficients				Correlation coefficients			
	Axis 1	Axis 2	Axis 3	Axis 4	Axis 1	Axis 2	Axis 3	Axis 4
Tree logging	-0.417	-0.1805	0.067	-0.2416	-0.5256	-0.3846	0.1458	-0.5062
Grazing	-0.3507	-0.0286	0.0669	0.1599	-0.4421	-0.0609	0.1456	0.335
Fire occurrence	-0.4997	0.1597	-0.2578	-0.0392	-0.6299	0.3403	-0.5611	-0.0821
Soil submersion	0.1834	-0.0148	0.0859	-0.0133	0.2311	-0.0315	0.187	-0.0279
Topography	0.4968	-0.1803	0.2514	-0.1583	0.6262	-0.3841	0.5471	-0.3317
Canopy cover density	0.2328	-0.0838	0.1078	-0.106	0.2934	-0.1785	0.2346	-0.2221
Soil types	0.2261	-0.0222	0.0404	-0.0524	0.2849	-0.0473	0.0879	-0.1097
Protection status	0.4584	0.3027	-0.1932	-0.3181	0.5777	0.6449	-0.4206	-0.6665

Appendix 2. Summary of the statistical outputs from DCA for all the 75 relevés (see Figure 4.2)

Axes	1	2	3	4	Total inertia
Eigenvalues	0.605	0.378	0.264	0.216	7.796
Lengths of gradient	4.671	4.425	3.297	2.631	
Species-environment correlations	0.793	0.469	0.459	0.477	
Cumulative percentage variance of species data	7.8	12.6	16	18.8	
Cumulative percentage variance of species-environment relation	26.8	33.2	0	0	
Sum of all eigenvalues					7.796
Sum of all canonical eigenvalues					1.188

Appendix 3. Canonical and correlation coefficients of the first four axes of the DCA for UPA (see Figure 4.3)

Environmental variables	Canonical coefficients				Correlation coefficients			
	Axis 1	Axis 2	Axis 3	Axis 4	Axis 1	Axis 2	Axis 3	Axis 4
TN10	-0.3823	-0.0006	-0.0202	-0.0885	-0.4353	-0.0008	-0.0263	-0.1137
TN20	-0.2005	-0.0745	-0.1052	0.022	-0.2283	-0.1	-0.1367	0.0283
SOC10	-0.0197	0.1184	-0.0114	-0.2173	-0.0224	0.1589	-0.0148	-0.2792
SOC20	-0.1343	0.1973	-0.0384	-0.081	-0.153	0.2649	-0.0499	-0.1041
pH10	-0.1416	0.2197	0.0638	0.1663	-0.1612	0.295	0.0829	0.2137
pH20	-0.1186	-0.054	-0.3578	0.1926	-0.135	-0.0725	-0.4649	0.2475
Tree logging	0.3115	-0.4112	0.09	0.1307	0.3547	-0.5521	0.117	0.168
Grazing	0.3854	0.3406	0.1321	0.1666	0.4388	0.4573	0.1717	0.2141
Fire occurrence	0.38	-0.2805	0.1447	-0.115	0.4327	-0.3765	0.188	-0.1478
Soil submersion	-0.5359	0.0261	0.1034	-0.2741	-0.6103	0.0351	0.1343	-0.3522
Topography	-0.6267	-0.1755	-0.0618	0.0429	-0.7136	-0.2356	-0.0803	0.0551
Canopy cover density	-0.4164	0.002	-0.2816	-0.0218	-0.4741	0.0026	-0.3659	-0.028
Soil texture	-0.1482	0.186	0.1528	-0.0129	-0.1688	0.2497	0.1985	-0.0166
TWI	-0.1778	-0.1716	0.1224	-0.0921	-0.2024	-0.2304	0.159	-0.1183
Altitude above channel level	0.1633	-0.0762	0.0522	0.2122	0.1859	-0.1023	0.0678	0.2727
Slope	0.1868	0.0944	-0.3683	-0.0215	0.2127	0.1268	-0.4785	-0.0276

Appendix 4. Summary of the statistical outputs from the DCA ordination in 36 relevés of UPA (see Figure 4.3)

Axes	1	2	3	4	Total inertia
Eigenvalues	0.4	0.185	0.136	0.101	2.936
Lengths of gradient	3.253	1.853	2.121	2.009	
Species-environment correlations	0.878	0.745	0.77	0.778	
Cumulative percentage variance of species data	13.6	19.9	24.5	28	
Cumulative percentage variance of species-environment relation	18.9	26.3	0	0	
Sum of all eigenvalues					2.936
Sum of all canonical eigenvalues					1.483

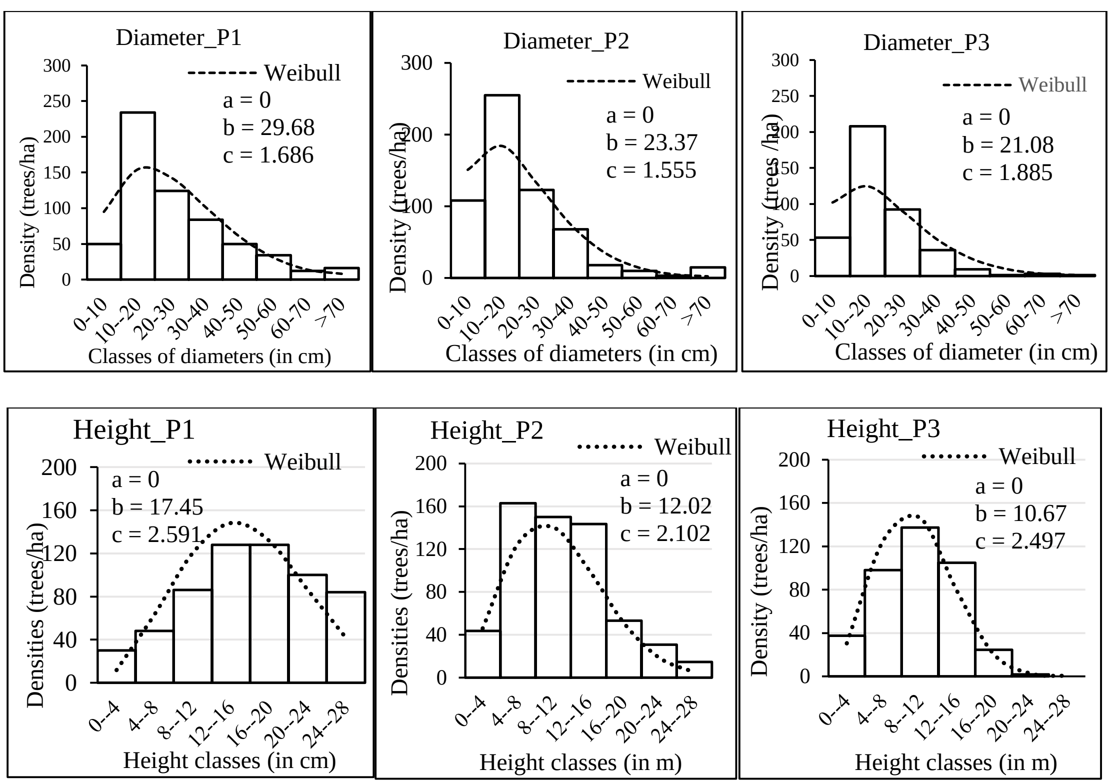
Appendix 5. Canonical and correlation coefficients of the first four axes of the DCA in PA (See Figure 4.5)

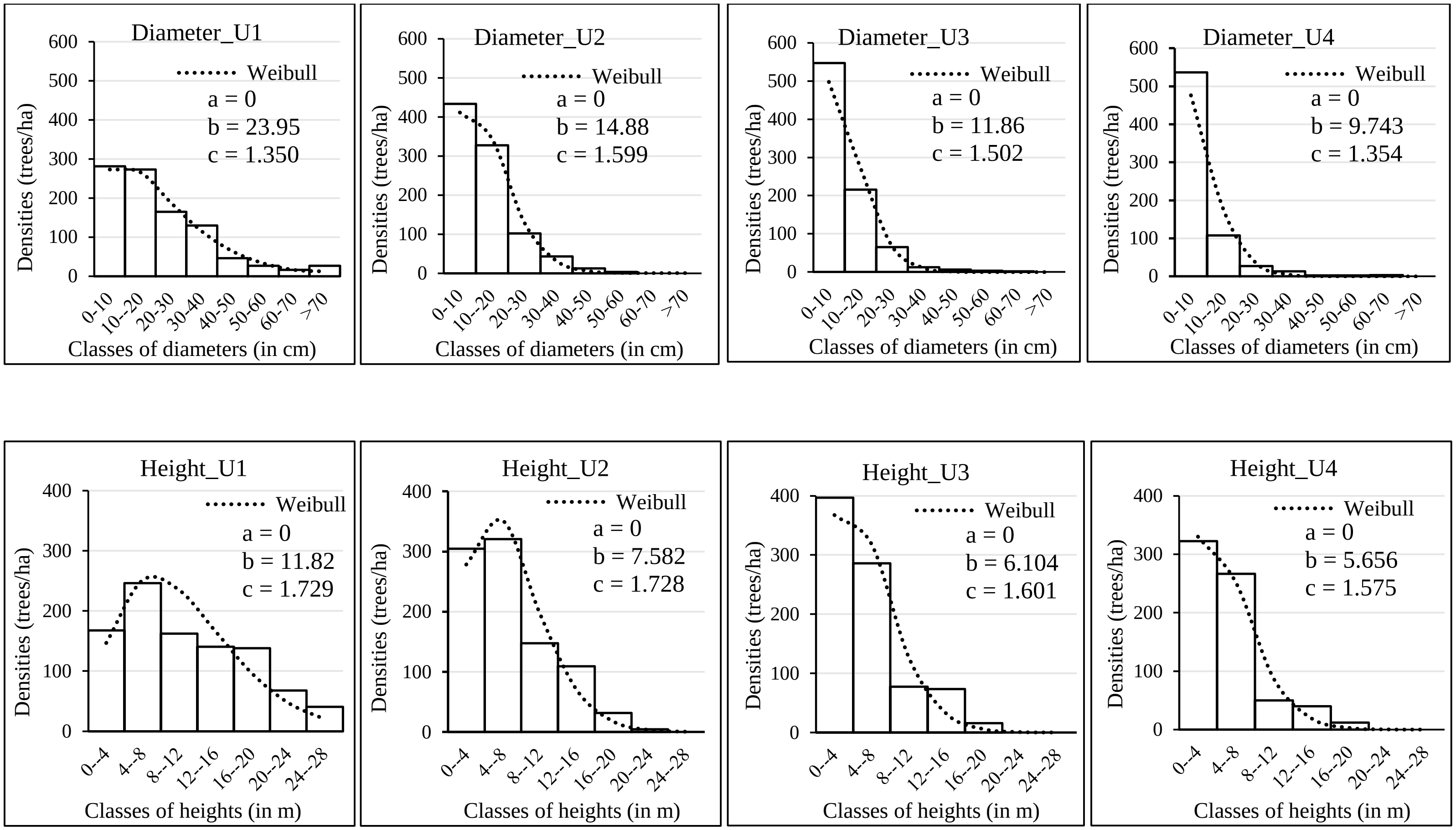
Environmental variables	Canonical coefficients				Correlation coefficients			
	Axis 1	Axis 2	Axis 3	Axis 4	Axis 1	Axis 2	Axis 3	Axis 4
TN10	0.6874	-0.0752	0.1454	0.0638	0.7881	-0.0829	0.1874	0.0945
TN20	0.4906	-0.1964	0.2397	-0.0845	0.5625	-0.2164	0.309	-0.1252
SOC10	0.5062	-0.0377	0.1883	-0.038	0.5803	-0.0416	0.2426	-0.0564
SOC20	-0.1262	-0.3527	0.0509	-0.0062	-0.1446	-0.3887	0.0656	-0.0091
pH10	-0.5329	-0.2294	-0.1681	0.1003	-0.611	-0.2529	-0.2167	0.1486
pH20	-0.3223	-0.113	0.2022	0.1873	-0.3695	-0.1245	0.2606	0.2775
Tree_logging	-0.2674	-0.4294	-0.1894	-0.4517	-0.3065	-0.4733	-0.2441	-0.6691
Grazing	-0.2522	-0.0124	-0.1324	-0.0754	-0.2891	-0.0137	-0.1706	-0.1117
Fire_occurrence	-0.5404	0.1874	-0.401	-0.1054	-0.6195	0.2065	-0.5168	-0.1561
Soil submersion	-0.0198	-0.0465	0.2045	0.2938	-0.0228	-0.0513	0.2635	0.4353
Topography	0.5349	-0.3255	0.3962	0.0918	0.6132	-0.3588	0.5105	0.136
Canopy cover density	-0.0205	-0.5024	0.159	0.0157	-0.0235	-0.5537	0.2049	0.0232
Soil_texture	0.186	0.174	0.3206	0.0487	0.2132	0.1918	0.4131	0.0721
TWI	0.3491	-0.1376	0.4182	-0.0801	0.4002	-0.1517	0.539	-0.1186
Altitude above channel level	-0.0601	0.1984	-0.2705	0.1319	-0.0689	0.2187	-0.3486	0.1954
Slope	-0.1598	-0.1526	-0.0505	-0.2194	-0.1832	-0.1682	-0.0651	-0.325

Appendix 6. Summary of the statistical outputs from the DCA for 36 relevés in PA (see Figure 4.5)

Axes	1	2	3	4	Total inertia
Eigenvalues	0.642	0.389	0.263	0.201	7.201
Lengths of gradient	4.19	3.348	2.792	2.81	
Species-environment correlations	0.872	0.907	0.776	0.675	
Cumulative percentage variance of species data	8.9	14.3	18	20.8	
Cumulative percentage variance of species-environment relation	12.3	21.4	0	0	
Sum of all eigenvalues					7.201
Sum of all canonical eigenvalues					3.587

Appendix 7. Distribution of tree diameter and height in the discriminated plant communities





Note: P1, P2 and P3 refer to the three described plant communities in PA. U1, U2, U3 and U4 represent the four plant communities in UPA. The letters' values a (location parameter), b (scale parameter) and c (shape parameter or Weibull slope) are the three parameters of the Weibull distribution function modelled for each vegetation group.

Appendix 8. Separability tests of Jeffries-Matusita (in bold) and of Transformed Divergence (Light)

Year		Forests	Woodlands	Savannahs	Croplands	Built-up	Water
1972	Forests		1.23	1.75	1.99	1.99	2.00
	Woodlands	**1.64**		1.07	1.95	1.91	2.00
	Savannahs	**1.99**	**1.55**		1.77	1.61	2.00
	Croplands	**2.00**	**2.00**	**1.98**		0.89	2.00
	Built-up	**2.00**	**2.00**	**1.94**	**1.04**		2.00
	Water	**2.00**	**2.00**	**2.00**	**2.00**	**2.00**	
1987	Forests		1.93	1.99	1.99	1.99	1.99
	Woodlands	**2.00**		1.46	1.86	1.99	1.99
	Savannahs	**2.00**	**1.85**		1.76	1.96	1.99
	Croplands	**2.00**	**1.99**	**1.99**		1.61	1.99
	Built-up	**2.00**	**2.00**	**1.99**	**1.90**		1.99
	Water	**2.00**	**2.00**	**2.00**	**2.00**	**2.00**	
2000	Forests		1.31	1.95	1.99	1.99	2.00
	Woodlands	**1.32**		1.73	1.96	1.99	2.00
	Savannahs	**1.99**	**1.95**		1.12	1.77	2.00
	Croplands	**1.99**	**1.99**	**1.22**		1.73	2.00
	Built-up	**2.00**	**2.00**	**1.90**	**1.82**		2.00
	Water	**2.00**	**2.00**	**2.00**	**2.00**	**2.00**	
2014	Forests		1.50	1.90	1.99	2.00	2.00
	Woodlands	**1.71**		1.76	1.99	2.00	2.00
	Savannahs	**1.99**	**1.98**		1.99	2.00	2.00
	Croplands	**2.00**	**2.00**	**1.99**		1.99	2.00
	Built-up	**2.00**	**2.00**	**2.00**	**1.99**		1.99
	Water	**2.00**	**2.00**	**2.00**	**2.00**	**2.00**	

Appendix 9. Accuracy assessment reports of the produced land cover maps from Landsat archives

			Ground truth (pixels)							Accuracy assessment		
		Land cover types	Forests	Woodl	Sav	Cropl	Built areas	Water	Total (Pixel)	Prod Acc (%)	User Acc (%)	Ov. acc. (Kappa coef.)
Classification (pixels)	1972	Forests	94	9	0	0	0	0	103	86.24	91.26	**68.86 (0.63)**
		Woodlands	13	94	6	2	0	0	115	85.45	81.74	
		Savannahs	2	7	95	76	33	0	213	82.61	44.6	
		Croplands	0	0	7	20	47	0	74	19.61	27.03	
		Built areas	0	0	7	4	55	0	66	40.74	83.33	
		Water	0	0	0	0	0	113	113	100	100	
		Total (Pixels)	109	110	115	102	135	113	684	414.65	427.96	
	1987	Forests	**45**	3	0	0	0	0	48	81.82	93.75	**91.32 (0.79)**
		Woodlands	7	**32**	5	9	0	1	54	61.54	59.26	
		Savannahs	3	17	**53**	20	0	5	98	75.71	54.08	
		Croplands	0	0	12	**15**	0	0	27	33.33	55.56	
		Built areas	0	0	0	1	**10**	2	13	100	76.92	
		Water	0	0	0	0	0	**739**	739	98.93	100	
		Total (Pixels)	55	52	70	45	10	747	979			
	2000	Forests	**24**	3	0	0	0	0	27	92.31	88.89	**90.66 (0.88)**
		Woodlands	2	**21**	4	0	3	2	32	72.41	65.63	
		Savannahs	0	5	**33**	5	3	2	48	84.62	68.75	
		Croplands	0	0	2	**24**	0	0	26	82.76	92.31	
		Built areas	0	0	0	0	**125**	0	125	95.42	100	
		Water	0	0	0	0	0	**74**	74	94.87	100	
		Total (Pixels)	26	29	39	29	131	78	332			
	2014	Forests	**47**	1	1	0	0	0	49	79.66	95.92	**91.88 (0.89)**
		Woodlands	8	**38**	8	1	0	0	55	84.44	69.09	
		Savannahs	4	6	**35**	8	0	0	53	71.43	66.04	
		Croplands	0	0	5	**121**	20	0	146	91.67	82.88	
		Built areas	0	0	0	2	**221**	0	223	91.7	99.1	
		Water	0	0	0	0	0	**262**	262	100	100	
		Total (Pixels)	59	45	49	132	241	262	788			

Appendix 10. Detailed pairwise transition matrices for the four periods

A.From 1972 to 1987

		To 1987						
		Forests	Woodlands	Savannahs	Croplands	Settlements	Water	**Total loss**
From 1972	Forests	**2.05**	3.22	0.36	0.01	0.00	0.00	**3.60**
	Woodlands	3.54	**40.88**	18.89	0.50	0.02	0.01	**22.96**
	Savannahs	0.59	15.78	**12.97**	0.84	0.09	0.01	**17.31**
	Croplands	0.00	0.07	0.09	**0.01**	0.00	0.00	**0.17**
	Settlements	0.00	0.01	0.02	0.00	**0.00**	0.00	**0.03**
	Water	0.00	0.01	0.01	0.00	0.00	**0.00**	**0.02**
	Total gain	**4.13**	**19.09**	**19.37**	**0.14**	**0.11**	**0.02**	

B.From 1987 to 2000

		To 2000						
		Forests	Woodlands	Savannahs	Croplands	Settlements	Water	**Total loss**
From 1987	Forests	**3.22**	2.52	0.42	0.01	0.00	0.00	**2.95**
	Woodlands	4.77	**36.81**	17.25	1.09	0.07	0.00	**23.18**
	Savannahs	0.52	14.92	**15.01**	1.74	0.15	0.00	**17.33**
	Croplands	0.01	0.33	0.85	**0.15**	0.02	0.00	**1.21**
	Settlements	0.00	0.02	0.06	0.02	**0.02**	0.00	**0.10**
	Water	0.00	0.01	0.01	0.00	0.00	**0.00**	**0.02**
	Total gain	**5.30**	**17.80**	**18.59**	**2.86**	**0.24**	**0.00**	

C.From 2000 to 2014

		To 2014						
		Forests	Woodlands	Savannahs	Croplands	Settlements	Water	**Total loss**
From 2000	Forests	**5.08**	2.04	1.34	0.06	0.01	0.00	**3.45**
	Woodlands	4.12	**21.12**	27.03	2.26	0.07	0.01	**33.49**
	Savannahs	1.75	4.01	**23.11**	4.58	0.16	0.01	**10.51**
	Croplands	0.05	0.11	1.61	**1.21**	0.03	0.00	**1.80**
	Settlements	0.03	0.01	0.09	0.09	**0.03**	0.00	**0.22**
	Water	0.00	0.00	0.00	0.00	0.00	**0.00**	**0.00**
	Total	**5.95**	**6.17**	**30.07**	**6.99**	**0.27**	**0.02**	

D.From 1972 to 2014

		To 2014						
		Forests	Woodlands	Savannahs	Croplands	Settlements	Water	**Total loss**
From 1972	Forests	**2.59**	1.18	1.76	0.11	0.00	0.00	**3.05**
	Woodlands	5.90	**20.93**	33.58	3.35	0.08	0.01	**42.92**
	Savannahs	2.52	5.16	**17.73**	4.67	0.19	0.01	**12.46**
	Croplands	0.01	0.01	0.08	**0.07**	0.01	0.00	**0.11**
	Settlements	0.00	0.00	0.02	0.01	**0.00**	0.00	**0.03**
	Water	0.00	0.00	0.01	0.00	0.00	**0.00**	**0.01**
	Total	**8.43**	**6.35**	**37.45**	**8.14**	**0.29**	**0.02**	

Appendix 11. Sheet for field characterisation using factor scoring approach

Semi-qualitative approach for site-specific evaluation of soil erosion in the Mo basin

Note: This sheet is adapted from Tamene (2005) in order to evaluate on-site soil erosion / deposition processes in the Mo basin. Due to inaccessibility in some cases, the subunits of Tchamou and Bouale of the Mo basin were selected for field visits. Scores are affected to each factor based on expert judgement.

Expert name:……………………………. Date: ………………………

1. Site identification

GPS coordinates	Lat:………………	Long:……………..	Alt:……………….
Sub-unit:			
LUC unit[7]:			
Fire occurrence:			
Tree logging:			
Soil texture:		Photo numbers:	

2. Characteristics and factors of on-site erosion and off-site deposition

Hillslope dominant Attributes (on-site erosion)	Possible scores		
	3	2	1
Surface cover (density)	Poor	Medium	Good
Level of degradation (evidences of erosion)	High	Medium	Low
Position to streams/gullies	Near	Medium	Far
Available material for detachment (surface nature)	High	Medium	Low
Average slope steepness	Steep	Medium	Gentle
Presence and extent of depositional sites	Low	Medium	High
Presence and intensity of other disturbances[8]	High	Slight	None
Physical structure of soil particle	Coarse	Medium	Fine
Total			
Gully/stream dominant attribute (off-site delivery)			
Drainage network (density of gullies/streams)	High	Medium	Low
Status of gullies/streams (stability, collapse)[9]	Severe	Slight	None
Average slope of gully/stream path)	Steep	Medium	Gentle
Evidences of deposition at gully/stream floor[10]	Low	Medium	High
Degree of disturbance by livestock/cultivation[11]	High	Medium	Low
Conservation practices	None	Medium	High
Average distance to a stream line[12]	Near	Medium	Far
Total			

[7] LUC : 1. Dry forest, 2. Riparian forest, 3. Woodland, 4. Savannah, 5. Shrubs, 6. Farm, 6. Fallow.
[8] Presence of disturbances such as roads, pavement, stones, etc.
[9] Bank stability as well as gully potential enlargement.
[10] Deposition material at site, obstructing the flow of sediments
[11] Grazing and cultivation footprints at the site
[12] Near (< 100 m); Medium (100 – 300 m); Far (> 300 m)