College of Natural and Computational Sciences

Centre for Environmental Science

Impacts of Land Use and Land Cover Changes on Land Surface Temperature, Soil Erosion and Soil Quality Deterioration in Suha Watershed, Northwestern Highlands of Ethiopia.

Candidate: Nigussie Yeneneh

A Dissertation Submitted to

The Centre for Environmental Science

In Partial Fulfillment of the Requirements for the Degree of Doctor of Philosophy in Environmental Science

Addis Ababa University

Addis Ababa, Ethiopia

July, 2024

ADDIS ABABA UNIVERSITY
GRADUATE PROGRAMS

This is to certify that the dissertation prepared by **Nigussie Yeneneh Gelaw**, entitled **Impacts of Land Use and Land Cover Changes on Land Surface Temperature, Soil Erosion and Soil Quality Deterioration in Suha Watershed, Northwestern Highlands of Ethiopia** and submitted in *Partial* fulfillment of the requirements for the Degree of Doctor of Philosophy (Environmental Science) complies with the regulations of the University and meets the accepted standards with respect to originality and quality.

Signed by the Examining Committee:

Name	Signature	Date
1. Advisor: Prof. Eyasu Elias		1/8/24
4. Internal Examiner: Dr. Berhanu Tamiru		
5. External Examiner: Dr Temesgen Desalegn		01 August 2024
6. Chairman: Prof. Ahmed Hussen		

July 30, 2024

ABSTRACT

Impacts of Land Use/Land Cover Changes on Land Surface Temperature, Soil Erosion and Soil Quality Deterioration in Suha Watershed, Northwestern Highlands of Ethiopia

Suha watershed found in the Upper Blue Nile Basin of Ethiopia is facing severe soil erosion and soil quality degradation problems due to its biophysical and socioeconomic characteristics. However, there is a lack of comprehensive data on the land use land cover changes and their impacts on land surface temperature, soil erosion risk and sediment yield, soil quality indicators as well as soil nutrient flows and balances in this watershed. Therefore, this study was undertaken to analyze the spatiotemporal changes in land use and land cover and to examine their impacts on the land surface temperature, soil erosion and sediment export, soil quality indicators and soil nutrient flows in the Suha watershed, northwestern highlands of Ethiopia. Multi-temporal Landsat images covering the periods from 1985 to 2019 were utilized to examine changes in land use and land cover (LULC) as well as land surface temperature (LST) variations with the application of GIS and remote sensing techniques. The process involved image preprocessing, supervised classification, accuracy assessment, and change detection. Quantification of soil loss and sediment export was carried out using the Revised Universal Soil Loss Equation (RUSLE) integrated with ArcGIS. Various data inputs, including digital elevation model, LULC, precipitation, soil types, and conservation strategies were used for this purpose. To evaluate the status of soil quality indicators, a total of 27 composite surface soil samples (0-30 cm) were collected from adjacently located land-use systems in three replications from each elevation gradient of the watershed. Standard procedures were applied to analyze selected physical and chemical soil quality parameters. The study also investigated soil nutrient depletion by quantifying nutrient flows and balances for macronutrients (N and P) in different elevation gradient of the watershed. Significant changes in land use and land cover were observed in the Suha watershed over the past 35 years. Agricultural land has increased by 15,418 hectares for the entire period at the expense of grazing and shrub lands. Barren land also expanded significantly due to cultivation of marginal and steep slope areas and poor land management. The study also revealed spatial and temporal variations in Land Surface Temperature (LST), with impermeable surfaces having the highest values. The average annual soil loss rate in this watershed varied over the years, with rates of 15, 22, 31, and 30 tons per hectare per year for the periods 1985, 1999, 2009, and 2019 respectively. The mean annual sediment yield was 4, 6, 8, and 8 tons per hectare per year for the same periods. Soil loss varied across different land use and land cover categories and landscape positions, with the highest mean value observed in bare land (54-103 t/ha/yr), followed by cultivated land (17-29 t/ha/yr). About 32% (25,660 ha) of the watershed experienced severe and very severe soil erosion in the past 35 years. The analysis of variance results of soil quality indicators showed significant changes in selected soil quality indicators across different land use systems. Specifically, the index of soil aggregate stability (ISS), organic carbon (OC), total nitrogen (TN), and C: N ratio revealed significant decreases in the cultivated land use system compared to other land use systems. Conversely, the content of available phosphorus (AP) was significantly higher in cultivated land. Moreover, the aggregated soil quality index (SQI) values have shown that soil quality is rated as low in cultivated fields, optimal in grazing land and high in forest land. The results of soil nutrient balances indicated that the rate of nutrient depletion is higher in the highland areas than the midland areas. N and P balances were -77.1 kg N /ha/yr and -11.9 kg P/ha/year for the wheat farming system (highland area), which are rated as one of the highest in Africa. For teff farming system, the values were -39.3 kg N/ha/yr and -1.4 kg P/ha/yr. The major negative drivers are removals in the harvested products (OUT1) and crop residues (OUT2) followed by soil erosion (OUT5). Overall, these studies provide valuable insights into the changes in land use/land cover, land surface temperature, soil erosion risk, soil quality and nutrient balances, contributing to a better understanding of the environmental dynamics. Sub watersheds that are found on the steep slope landscapes and are experiencing severe soil erosion that demand immediate interventions. effective soil and water management strategies should be established and integrated with vegetative measures to provide multiple benefits to the farmers. In order to reduce the volume of crop residue removal as household energy (OUT2), promoting energy saving technologies such as the use of fuel stoves, saving stoves and biogas is crucial.

Keywords: LULC change, LST, GIS & remote sensing, Landsat image, RUSLE, Soil erosion, Soil quality and Soil nutrient flows

Nigussie Yeneneh

Addis Ababa University, 2024

ACKNOWLEDGMENT

This research wok was found in its present form as a result of supports obtained from different professionals and institutions. I would like to extend my profound and special sincere thanks and appreciations to my esteemed advisor, Prof. Eyasu Elias, who efficiently guided me throughout my research work. He has invested his precious time and deep knowledge in various stages of my work starting from proposal development to final write-up of the thesis. His unreserved support, supervision, encouragement and patience have positively and significantly impacted on the completion of this thesis work. I would like to express my sincere gratitude to Dr. Gudina Legese, my co-advisor, for his motivation, guidance and supervision, critical suggestions and comments during data analysis (particularly for his technical guidance on the applications of GIS and Remote sensing part), unreserved support, and encouragement in the course of my study. Above all, I would like to express my special thanks to my advisors for their financial support for laboratory costs, which I was in trouble in this regard and their fruitful encouragement while I have been facing problems during my work.

I am grateful to the Center for Environmental Science and PhD program coordinator for their follow up, encouragement and updating information during the course of the study period. All colleagues and PhD candidates of the Center for Environmental Science are also acknowledged for their support and motivation during my study. I am also delighted to acknowledge Addis Ababa University, Debre Markos University and Ministry of Agriculture for their financial support; Ethiopian Construction Design & Supervision Works Corporation for allowing us to analyze soil samples in their laboratory; National Meteorological Agency for providing time series climate data. I am very grateful to Simeneh Gedefaw and Alene Goshu for their technical support for GIS and remote sensing. Kind cooperation of development agents during our field work and data collection was highly appreciated. Farmers of the two villages (Yekebehana and Telemt) who allowed us to collect data from their farm land and giving all necessary information for our work are also acknowledged. Special thanks also go to my family members for their holistic support and encouragement while I faced problems.

DEDICATION

This PhD dissertation is dedicated to my parents.

Table of content

Content **Page**

List of figures

Figure **page**

List of tables

Table **Page**

List of appendices..…………………………………………………………………...…162

ACRONYMS

ANOVA	Analysis of variance
AP	Available phosphorus
ASTER	Advanced Space borne Thermal Emission and Reflection Radiometer
AWC	Available water content
BT	Brightness Temperature
CEC	Cation Exchange Capacity
C: N	Carbon to nitrogen ratio
ENVI	Environment for Visualization Images
DAP	Di-Ammonium Phosphate
DEM	Digital Elevation Model
DN	Digital number
ESP	Exchangeable sodium percentage
ETM+	Enhanced Thematic Maper plus
FAO	Food and Agricultural Organization
FC	Field capacity
GIS	Geographic Information System
GPS	Global Position System
IDW	Inverse Distance Weight
ISFM	Integrated Soil Fertility Management
ISS	Index of Soil aggregate Stability-
LULC	Land use and land cover
LSE	Land surface emissivity
LST	Land surface temperature
MoARD	Ministry of Agriculture and Rural Development
MoWR	Ministry of Water Resource
NDVI	Normalized Difference Vegetation Index
OLI	Operational Land Imager

OM	Organic matter
PA	Producer’s accuracy
PBS	Percent base saturation
PV	Proportion of Vegetation
PWP	Permanent wilting point
RF	Rain fall
ROI	Region of Interest
RUSLE	Revised Universal Soil Loss Equation
SAS	Statistical Analysis Software
SCRP	Soil Conservation Research Project
SDR	Sediment delivery ratio
SE	Standard error
SOC	Soil organic carbon
SQDI	Soil quality deterioration index
SQI	Soil quality index
SQR	Soil quality rating
SSA	Sub-Sahara Africa
SVM	Support vector machine
SY	Sediment yield
TIRS	Thermal Infra Red Sensor
TM	Thematic Maper
TN	Total nitrogen
TOA	Top of the atmosphere
UA	User’s accuracy
USGS	United States Geological Survey
UTM	Universal Transverse Mercator

CHAPTER ONE: GENERAL INTRODUCTION

1.1. Background and Justification

Land degradation and its negative impacts, which are primarily driven by land use and land cover (LULC) changes, are becoming central issues globally (Li et al., 2020). In Africa, the dynamics of land use and land cover changes are mainly caused by the ever-increasing population and associated anthropogenic activities (Eva et al., 2006). Land-use change signifies the transition of an area of land for a specified purpose and is induced by anthropogenic activities, whereas land-cover change describes the features of the land surface (Patel et al., 2019). Land use and land cover change is driven by a combination of proximate and underlying causes, with the former being related to anthropogenic activities and the latter to policy and socioeconomic factors such as poverty and market incentives. In the Ethiopian highlands, LLULC changes cause soil degradation, and soil quality deterioration, thereby threatening the suitability of the agroecosystems. In addition to deforestation, major LULC changes have occurred in all land use types at the national and local levels. A study conducted by Hassen and Assen (2018) in the Gelda catchment, northwestern highlands of Ethiopia, revealed that cultivated land has increased by 58% at the expense of natural forest and grasslands over the past 57 years (between 1957 and 2014). Similarly, Fentie et al. (2020) reported a significant expansion of agricultural land at the expense of grazing land, bush and shrub lands and through the conversion of the natural forest areas to other land-use systems.

Previous studies have shown that land use and land cover changes result in negative impacts from local to global scales. Li et al. (2020) reported the impacts of LULC changes on the global environment (e.g., climate change) and their implications for sustainable development. Hassen and Assen (2018) found that deforestation, soil erosion, soil quality degradation, and water scarcity are among the land degradation problems related to LULC changes. Impacts on ecosystem services such as loss of biodiversity and habitat quality were reported by Dinka and Chaka (2019); Patel et al. (2019). Other studies highlighted the effects of LULC changes on the local hydrological cycles (Damtea et al., 2020). Impacts on the land surface temperature were pointed out by (Kikon et al., 2016; Pal and Ziaul, 2017). All these reports point to the need for the detection and analysis of LULC change and managing its negative impacts for transitioning to environmental sustainability (Alam et al., 2020; Patel et al., 2019).

Soil erosion is a major challenge worldwide, having negative impacts on social, economic, and environmental development (Marques, 2021). According to research reports (Opeyemi et al., 2019), soil erosion accounts for about 84% of global land degradation, with annual soil loss ranging from 25 to 400 billion tons per year. Sub-Saharan Africa is the worst in the world; on the other hand, the livelihood of most population in these countries is highly dependent on their natural resource base (Nkonya et al., 2015). In these regions, soil erosion has threatened the livelihood of the population, because 67% of the land resource has been degraded in different severity classes (light to severely degraded classes) (Sileshi et al., 2019). The highlands of Ethiopia are among the most degraded regions in sub-Saharan Africa (SSA) (Fenta et al., 2021). The Ethiopian highlands reclamation study (EHRS) estimated an annual soil loss of 42 tons/year/ha (FAO, 1986). Fenta et al. (2021) also documented the soil loss rates in different River Basins of Ethiopia and confirmed that the highest annual soil loss (573 million tons/year) was from the Blue Nile (Abay) Basin followed by the Tekeze basin (270.6 million tons/year). The severe soil erosion in these two basins was related to the rugged topography, deforestation driven by high population pressure, and poor land use practices (Kebede et al., 2021; Yesuph and Dagnew, 2019). A catchment level study confirmed that catchments found in the upper part of the Blue Nile are heavily affected by soil erosion (Berihun et al., 2020). The above findings confirmed that the severity of soil erosion and sediment export significantly vary across regions due to the difference in environmental, climatic and anthropogenic factors (Garcia-Ruiz et al., 2015).

Soil erosion is the major cause of several economic and environmental problems. Globally, it costs the international community about $400 billion (Opeyemi et al., 2019). In particular, the effects of soil erosion on soil fertility and food security have been widely reported (Marques, 2021). In Sub-Saharan Africa (SSA), soil erosion is the major factor for the degradation of more than 65% of agricultural land and subsequent loss of agricultural yield, food security and human wellbeing (Tully et al., 2015; Vlek et al., 2008). The removal of fertile soil and reduction of agricultural productivity is more severe in the highlands of Ethiopia as compared to Sub-Saharan Africa (SSA) due to excessive soil erosion (Dibaba et al., 2021; Hurni et al., 2015). Hurni (1993) reported that the productivity of soil in the highlands of Ethiopian is reducing at a rate of 2-3% annually due to soil erosion. Yesuf et al. (2007) also estimated soil erosion cost in terms of the annual GDP of Ethiopia and reported that GDP is reduced from 2% to 6.75% annually. Among the ecosystem services, loss of hydrological regulation that results in contamination of water bodies and silting up of dams in the downstream areas affects millions of rural and urban communities (Fenta et al., 2021). The storage capacity and life expectancy of reservoirs and lakes in Ethiopia have progressively been reduced due to the continuous deposition of sediment load from the up streams (Haregeweyn et al., 2017). For instance, the study by Zemadim et al. (2014) showed that 3.5 million m^3 of sediment has accumulated in the Koka dam in 23 years. Yitaferu (2007) also estimated siltation in Lake Tana and showed a sediment deposition rate of 14.84 million tons per year. The complete drying up of Lake Haramaya, which is one of the upland lakes in the Eastern plateau was partly due to silt deposition from the surrounding agricultural fields (Eshetu et al., 2014**)** and sediment accumulation in Cheleleka wetland in Ethiopia's Central Rift Valley was due to excessive soil erosion and subsequent deposition (Degife et al., 2019). The problems of siltation and nutrient enrichment in the Gilgel Gibe-I hydro-power dam were also reported by Devi et al. (2008).

Deterioration of soil physical, chemical and biological properties has become a global environmental problem, including nutrient depletion, reduced plant rooting depth, soil acidity and associated problems such as high phosphorus fixation, aluminum toxicity, low CEC and salinity (Zingore et al., 2015). Estimates of nutrient balances for the Ethiopian highlands indicated high annual nutrients loss (Teferi et al., 2016). The study of Zingore et al. (2015) revealed that the negative nutrient balances of macro nutrients (40-70 kg N/ha, 7-10 kg P/ha and 33-50 kg K/ha) were rated as one of the highest in SSA. This is in direct contrast with the SSA level nutrient balance of -41 kg N, -6 kg P and -26 K per ha (Stoorvogel and Smaling, 1990). Soil fertility is a key factor that controls agricultural productivity (Elias, 2004). However, soil erosion, soil quality deterioration and nutrient depletion are major problems observed in the north western highlands of Ethiopia including the study watershed. Understanding the dynamics of land use/cover change and soil erosion risk is vital for designing appropriate management strategies; improving soil health and ensuring agricultural sustainability. Moreover, the knowledge of soil quality and nutrient flows is crucial for applying location specific management strategies. However, there is limited information on the extent to which these problems are severe in this particular study area (Teferi et al., 2013; Simane et al., 2013) and this study was conducted to fill these gaps. Therefore, the overall objective of this study was to analyze the land use/cover changes and evaluate their impacts on land surface temperature, soil erosion, and soil quality including soil nutrient balances for macro-nutrients (N and P) in Suha watershed of the Upper catchment of the Blue Nile basin.

1.2. Problem statement

LULC changes have been identified as the main driver of soil degradation in the northern highlands of Ethiopia. Various studies conducted in different regions of the country have shown significant transitions in land use and land cover, which have consequently resulted in to soil degradation (Dibaba et al., 2021; Gashaw et al., 2021). Ota et al. (2018) have highlighted the negative impact of anthropogenic activities, such as the conversion of forest land into cultivated fields, on soil fertility and quality. LULC changes driven soil degradation is the typical feature of the northern highlands of Ethiopia. The Suha watershed is highly threatened by soil erosion and nutrient depletion. The watershed is one of the catchments in the Upper Blue Nile Basin that exports large volumes of sediment into the Nile River and potentially causes siltation in the Grand Ethiopian Renaissance Dam (GERD). Moreover, Simane et al. (2013) have pointed out that soil fertility issues are not yet resolved in the area and are expected to remain a major challenge in the future. Local-level studies conducted in the Upper Blue Nile Basin gave more emphasis on the dynamics of land use land cover change and their driving forces. The impacts of LULC changes on soil fertility and soil quality have not been adequately studied and documented. In addition, few studies were conducted in the rural catchments of Ethiopia (Alemu, 2019; Athick et al., 2019) pertaining the impacts of LULC changes on LST. In this regard, previous studies about land use land cover changes and their impacts on LST were lacking and this research was conducted to fill these knowledge gaps.

Selection and applications of soil and water conservation technologies are site-specific due to the variation in physiographic, socioeconomic and climatic factors among regions or catchments (FAO, 2017). Hence, assessment and quantification of land degradation is the first essential step to persuade the government and local communities to take action before it is too late. Identification of the hotspots of soil erosion in the sub-watersheds also provides crucial information to guide soil conservation interventions and resource allocations. However, the volume of soil loss by runoff water and its spatial distribution and sediment yield of the Suha watershed have not yet been quantified. In addition, quantifying the level and direction of soil nutrient balance and soil erosion rate is imperative to identify the major land use types responsible for the soil quality degradation. This is necessary to design effective soil and site-specific management strategies so as to ensure agricultural sustainability, improve ecosystem services and environmental protection (Nguemezi et al., 2020; Tellen and Yerima, 2018). However, unlike other parts of the country where studies explore the impacts of LULC change on soil quality were conducted, no such information exists for the study area. Moreover, the alteration of soil quality indicators is site-specific depending on the types of management systems and hence, local level studies are imperative to improve soil condition (Shukla et al., 2006). Estimating and identifying the magnitude and direction of soil nutrient flows and balances at the watershed scale is also important to know the level of soil nutrient depletion and to take corrective measures. The knowledge of soil fertility helps to design effective and site specific management strategies that enhance agricultural productivity and ensure food security. In this regard, data on soil nutrient flows, soil nutrient balances and management strategies applied at the farm level were lacking in the study area. Moreover, the total annual sediment export from the entire watershed was expected to be high, which indicates the export of substantial amounts of soil nutrients with sediment.

Hence, regular monitoring of LULC changes and knowing its implications on LST change; soil erosion risk and sediment yield; soil quality change is crucially important for better management and sustainable use of environmental resources. It also helps us to identify hotspot areas of the watershed that demand priority for interventions. Moreover, soil nutrient balance assessment helps to know the level of nutrient mining and the types of applied management strategies. In addition, it provides vital information for decision-makers and development organizations to select sets of soil and site-specific soil management strategies so as to increase agricultural productivity and improve the livelihoods of the rural community. Suha watershed was selected for this research because of its environmental problems related to continuous land use and land cover changes; which in turn brought negative impacts on other environmental issues, including LST change, soil erosion and soil quality deterioration. During our field work, we have also confirmed that this watershed was significantly degraded and potential landscapes are still highly vulnerable to further environmental degradation because of human-induced activities. Moreover, regional and district offices identified this watershed as one of the catchments that experienced natural resources degradation and demand detailed research and development interventions. Generally, the following points were the reasons for the selection of this watershed.

- This watershed represents other catchments found in the northwestern highlands of Ethiopia because of its biophysical features (climatic conditions, physiographic characteristics, and soil types) and socioeconomic settings
- This area has been considered as high potential area for agricultural production, but through time it became under pressure and the potentials of natural resources were degraded and productivity reduced (Zeleke and Hurni, 2001).
- It is characterized by undulating topography and high population pressure; which are the major causes of environmental resources degradation
- Environmental resources (soil, forest and water) are highly degraded due to proximity and underlining causes (population pressure, over exploitation of natural resources, and poor land use systems).
- The presence of high soil erosion risk and sediment yield that causes soil nutrient export and high replacement cost; which intern results in food insecurity at the household level.
- This watershed has the potential of producing a significant amount of sediment to the Nile River and finally causing siltation in the GERD.
- The presence of poor land use systems; which are the proximity causes for critical environmental issues.

1.3. Research objectives

1.3.1. General objective

The general objective of this study was to analyze the land use and land cover change and evaluate its impacts on land surface temperature, soil erosion and sediment yield, and soil quality including soil nutrient balances for macro-nutrients (N, and P) in the Suha watershed of the Upper catchment of the Blue Nile basin.

1.3.2. Specific objectives

- To analyze the spatio-temporal land use/cover and land surface temperature changes in the Suha watershed over the past 35 years;
- To quantify LULC change-induced soil loss and sediment yield at the watershed level;
- To evaluate the impacts of LULC changes on soil quality in terms of its physical and chemical fertility along the toposequence of the watershed

- To quantify soil nutrient flows and balances in contrasting agroecologies under cereal-based agroecosystems.

1.4. Research questions

Based on the above specific objectives, the following general research questions were addressed during the course of this study.

- ❖ What seems the trend of LULC and LST changes in Suha Watershed over the past 35 years?
- ❖ How land use and land cover changes and landscape positions affect the rate of soil loss and sediment yield in Suha Watershed?
- ❖ Is there a significant change in soil physical and chemical quality indicators along the toposequence of the watershed under different land use systems?
- ❖ What seems the rate of soil nutrient flows and balances in different agroecosystems at the watershed scale?

1.5. Significance of the study

The findings of this research provide vital information for watershed level land restoration including the identification of priority areas for restoration investment, sites of nutrient losses and accumulation and sustainable land use practices. Given that the study area is situated in the headwaters of the Great Ethiopian Renaissance Dam (GERD), the findings are relevant to control soile erosion and sediment export that are vital for controlling the siltation of GERD and other water bodies. Points of departure in this study were exploring land use land cover changes and subsequent soil erosion rates across different land use types and nutrient balances by means of empirical modelling and primary data collection. Therefore, the findings are relevant for future research, development planning and policy decision. There are a number of NGOs and development partners in the area such as, Sustainable Land Management programme (SLMP), Climate Action through Land Management (CALM), Integrated Soil Fertility Management(ISFM), and Agricultural Growth Program (AGP) that can use the results of this research as an input in developing plans and implementation. The outcomes of this study can also be used by other researchers as baseline information to conduct additional research on relevant themes. It also provides new knowledge for researchers and professionals better understand the dynamics of land use and land cover, as well as their consequences on LST, soil erosion and soil quality deterioration. Moreover, it helps to raise awareness among the local community about the problems of natural resources degradation and to get them involved in watershed development programs.

1.6. Structure of the dissertation

This PhD dissertation has 5 chapters**.** The first chapter explains the general background of the thesis. It presents an over view of the dynamics of land use land cover change and its associated impacts; with emphasis given to Sub-Saharan Africa and Ethiopia's condition. It also highlights the extent and rate of soil erosion, sediment yield and their casual factors; soil quality changes and soil nutrient flows and balances. Chapter 2 provides a literature review on land use land cover dynamics and their drivers; impacts of LULC change on LST, soil erosion and soil quality. It also gives an over view of soil nutrient flows and balances at different spatial scales and management strategies. Methodological approaches were also explained in this part. Chapter 3 deals about materials and methods section (descriptions of the study area; materials and methods used in this research). Chapter 4 presents the results and discussion for each research objective. Chapter 5 deals with the conclusion and recommendations part. It outlines major findings, their implications and suggested recommendations for identified problems.

CHAPTER TWO: LITERATURE REVIEW

2.1. Land use/ land cover changes and major land use types (LUTs)

2.1.1. Trends of land use / land cover changes in Ethiopia

Land use/land cover (LULC) change is the major factor for various environmental problems and understanding the trend of LULC change is crucial to develop sustainable management strategies (Gashaw et al., 2017). In Ethiopia, several studies of LULC changes were carried out at different spatial scales; from Basin to watershed levels (Teferi et al., 2016; Minta et al., 2018). Most of these studies indicated that, there has been significant expansion of cultivated land at the expense of natural forest and grazing areas. Beyond deforestation, there have been significant changes in land use and land cover across all land use categories at national and local levels. A study conducted by Hassen and Assen (2018) in the Gelda catchment, northwestern highlands of Ethiopia revealed that farmland land has increased significantly by 57.68% at the expense of natural forest and grasslands in 57 year period (between 1957 and 2014). The results of Fentie et al. (2020) demonstrated that forest and shrub land use systems are reduced due to an expansion of cultivated land. Dinka and Chaka (2019); Gedefaw et al. (2020) from their local-level studies demonstrated the occurrence of significant LULC changes. The findings of Gashaw et al. (2017) also revealed that the area of cultivated land has increased by 22.5% during 34 years period (195-2018) in the Upper Blue Nile Basin where as grass land, natural forest cover and shrub land decreased by 36.1%, 45% and 41.5% respectively for the same period. The decline in forest cover was believed to be due to the expansion of agricultural land which was also observed in the Upper Gilgel Abay watershed of the Blue Nile basin (Rientjes et al., 2011). The expansion of crop lands was even more remarkable in the Derekolli catchment which increased from 65% in 1957 to 71% in 1986. Conversely, shrub land has diminished at the rate of 1.6% and 0.31% per year between the 1957-1986 and 1986-2000 periods, respectively (Tegene, 2002). Dessie and Johan (2007) illustrated the decline of forest cover in the south central Rift Valley region from 16% in 1972 to 2.8% in 2000.

The unique feature of the Upper Blue Nile basin is the expansion of not only crop lands but also barren land, and wooded grassland over the past four decades (1973-2000). An increase in the water body was also basically due to the construction of various hydro electric dams in the basin including the Ethiopian Great Renascence dam (Gebremicael et al., 2013). A reduction of grassland cover (88%) from 1972 to 2007 periods was also observed in the northern Afar rangelands while Bush land cover and cultivated land increased more than threefold and eightfold, respectively (Tsegaye et al., 2010). Tekle and Hedlund (2000) who studied LU/LC change in Kalu District of North-eastern Ethiopia reported a reduction in coverage of scrublands, riverine vegetation and forests, and an increase in remaining open areas, settlements, floodplains, and water bodies between 1958 and 1986 periods. In Kalu District, the area under cultivation was almost unchanged. In the same manner, forest cover continuously decreased in Lake Hawassa between 1986 and 2011 periods while woodland cover increased.

However, the change in cropland, bare land and grassland were very small (Wolka et al., 2015). On the other hand, some studies previously conducted in the degraded parts of northern Ethiopia such as Bantider et al. (2011) in the eastern Escarpment of Wello; Mekonen and Gebreyesus (2011) in Medego watershed and Haregeweyn et al. (2015) in Enabered watershed revealed the improvement of vegetation cover due to community afforestation, land rehabilitation and integrated watershed management activities,

2.1.2. Drivers of LULC changes

The drivers of LU/LC changes are classified as proximate and underlying causes. Among the proximate causes include land conversion from natural to agricultural land use leading to overgrazing, and deforestation (Twisa and Buchroithner, 2019) and weak institutional setup (Dinka and Chaka, 2019). Among the underlying causes include familiar themes such as population pressure, poverty, and policy disincentives including insecurity of land holding.

In the case of Ethiopia, most LULC changes are driven by underlying factors mainly population pressure and poverty (Abate, 2011b; Gashaw et al., 2014). For example, increase in population pressure in the highlands since the mid to the turn of the 20th century had accelerated deforestation and intensified cultivation (Hurni et al., 2015). Yalew et al. (2016) also demonstrated that the major land use/cover change drivers in the Upper Blue Nile Basin, Jedeb catchment were ever increasing human and livestock populations and weak infrastructure development such as lack of roads, markets and water sources.

Besides these causes, institutional and policy factors were found the drivers for the reduction of forest and grassland and the increase of cultivated land, bare land and shrub land in Geleda catchment during the 1957-2014 periods (Esa et al., 2018). For example, land tenure insecurity results in deforestation and extractive livelihood strategies. Urbanization, industrialization, and institutional building were found the drivers of LULC changes as demonstrated around Mekelle city in the Tigray region of Ethiopia (Tahir et al., 2013). Conversely, small-scale agriculture, commercial logging and commercial farms were the causes of forest cover loss in the south-central Rift Valley region during the 1972-2000 periods (Dessie and Johan, 2007). Therefore, the major drivers of LULC changes in Ethiopia emanated mainly from population growth, which is manifested mostly through the expansion of cultivated lands, even in areas where cultivation is almost impossible, and urban expansions.

The LULC changes pose multi-dimensional impacts on local climatic and environmental systems. The event influences the temporal and spatial dynamics of environmental and ecological systems including greenhouse gas emissions and biodiversity losses (Barlow et al., 2016); soil erosion and sedimentation (Debie et al., 2019); water resources (Birhanu et al., 2019; Dinka and Chaka, 2019); and climate change (Brovkin et al., 2013).Moreover, LU/LC changes influence surface energy balance, which is the major underlying process for temperature change (Jain et al., 2017), as well as additional severe weather conditions, such as drought and flooding.

2.1.3. Land surface temperature (LST) change driven by LULC changes

Land surface temperature (LST) refers to the radiative land surface skin temperature, which is highly variable following the thermal properties of land surface features (Pal and Ziaul, 2017; Zhang et al., 2009). The change in LST is a regional climate response to global climate change and has research and development importance in agriculture, water resource, ecology, agro-meteorology, and climate change over large spatial and temporal scales (He et al., 2018; Zhang and He, 2013). Therefore, studies on the relationship between LST and LUCCs are essential to provide the basis for regional and national planning and environmental monitoring and management. Besides, it is also essential to identify factors which control the spatial-temporal variation of land surface temperature triggered by LU/LC change (Haylemariyam, 2018; Balew and Korme, 2020). For instance, Sahana et al. (2016) investigated the effects of land use change from natural forest to paddy rice and settlement and found an increase in land surface temperature in the Sundarban Biosphere Reserve of India. Besides land use/land cover, elevation and other topographic features can cause changes in the land surface temperature (He et al., 2018). In the Russian Altay Mountains, topographic features like elevation were found to be the main drivers of land surface temperature changes. He et al. (2018) have concluded that topographic features such as elevation, slope and aspect significantly affect spatial variation of LST in the mountainous areas in the north east China. In areas where there is modification of vegetation cover like in urban centers, there is a change in the thermal regime which further triggers an increase in the air temperature of the surrounding areas (Zhang et al., 2009).

2.2. Concepts of soil quality and the effect of LULC changes on soil quality

2.2.1.Conceptual issues

Soil quality refers to the capacity of soil to give its ecosystem services, within the natural and managed ecosystem boundaries, to sustain biological productivity, maintain and enhance water and air quality, and support human health and habitation (Bünemann et al., 2018; Andrews et al. 2004). Hence, it is more complex than water and air quality as it provides multiple ecosystem services/functions. The concept of soil quality has emerged during the last decades and is used to assess land or soil quality under various systems (Ezeaku, 2015). This parameter determines the capacity of soil to deliver a range of functions that support ecosystem services including human health and wellbeing (Corstanje et al., 2017). Soil quality is a dynamic interaction between various physical, chemical and biological soil properties, which are influenced by many external factors such as land use/ land management and socio-economic priorities; these parameters are involved in the critical functioning of the soil (Maurya et al., 2020).

Soil quality can be categorized as inherent and dynamic qualities (Karlen et al., 2008; Larson and Pierce 1994). Inherent soil quality is the natural capacity of soil to function in certain climatic conditions. These characteristics are permanent and do not change by management or land use. They are often used to evaluate the suitability of soils for specific uses. On the other hand, dynamic soil quality describes the response of soil functions to land use/land cover changes or specific management strategies (Braimoh and Vlek, 2006), and hence the dynamic soil quality is the focal point of soil sustainable land management assessment. Land use change/management will result in a net positive or negative impact on the quality of the soil. For example, land use/land cover changes from forest land to other land use systems results in soil erosion (Bewket and Teferi, 2009; Haregeweyn et al., 2015), increases compaction (decreases the content of soil organic matter, water holding capacity and nutrient availability(Bewket and Stroosnijder, 2003). These effects ultimately lead to negative changes in soil quality indicators from the perspective of sustainable land use systems.

Soil quality cannot be measured directly; rather it is determined based on soil quality indicators (soil physical, chemical and biological properties) (Zornoza et al., 2015). The changes of these indicators due to land use changes and management activities can help us to evaluate whether the soil quality is degraded, stable or improved (Bünemann et al., 2018; Huang et al., 2021). Quantitative determination of these indicators is also site-specific due to the difference in soil management strategies in various ecosystems (Zhao et al., 2021). The sustainability of ecosystem service could be ensured if and only if soil quality is improved and maintained (Delelegn et al., 2017).

2.2.1.1. Soil quality indicators

Soil quality evaluation enables researchers to get a thorough understanding of the soil; to explore its responses to LULC changes and to develop sustainable management strategies (Rahman et al., 2020). It cannot be measured directly. As a result, soil quality indicators are required to evaluate its status. Scientists consider a wide range of measurable physical, chemical and biological soil parameters as the quantitative indicators of soil quality like soil texture, aggregate stability, soil organic carbon (SOC), electrical conductivity (EC), total nitrogen, CEC, exchangeable bases, soil pH, soil microbial characterization etc. The assessment of soil quality by understanding the threshold range of these indicators in certain ecology also shows the sustainability potential of that soil. There are three main categories of soil quality indicators; which include: chemical, physical and biological. The selection of soil quality indicators needs to ensure that they are sensitive and responsive to land use changes/ land management strategies across space and time scales (Corstanje et al., 2017).

Soil physical quality indicators: refers to soil physical properties which include soil texture, soil depth, bulk density, aggregate stability, crusting, porosity and compaction. Physical indicators primarily reflect limitations for root growth, and water infiltration within the soil profile.

Soil chemical quality indicators: The soil's chemical condition affects soil-plant relations, water quality, buffering capacities, availability of nutrients and water to plants and other organisms, mobility of contaminants, and some physical conditions, such as the tendency for the crust to form. Chemical soil quality indicators include soil pH, soil salinity, nutrient availability (N, P, K, micro-nutrients), cation exchange capacity (CEC), exchangeable bases, and the concentrations of elements that may be potential contaminants (heavy metals, radioactive compounds, etc.).

Soil biological quality indicators: Biological soil quality includes a variety of soil functions associated with the living component of the soil ecosystem. Prominent biological soil quality indicators include the diversity and composition of species of soil organisms, particularly soil microorganisms and soil flora (the types of plants, their root systems and the vegetative litter produced at the soil surface). In this respect, the amount and quality of soil organic matter (SOM) or organic carbon (SOC) is a reflection of the nature and abundance of soil flora and fauna (Bajracharya et al., 2007). Hence, soil organic matter (SOM) content is widely considered a key indicator of soil biological quality indicator. Moreover, SOM has been found to be beneficial for nutrient retention/recycling, soil productivity, water-holding capacity and carbon sequestration (Seely et al., 2010; Six and Paustian, 2014). For soil quality evaluation, integration of soil physical, chemical and biological properties and establishing soil quality index is required. The result of SQI provides a single numerical value by integrating the values of measured soil properties.

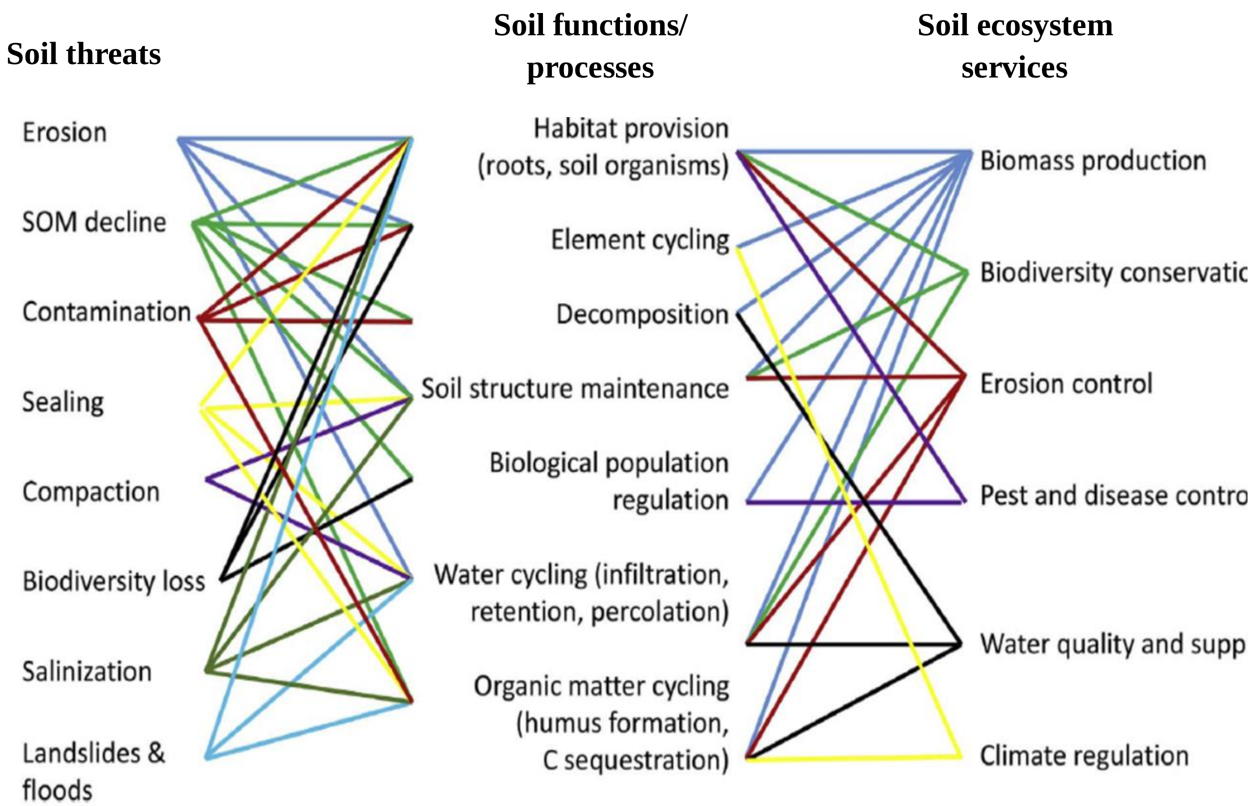


Fig 2.1. The relationship between soil threats, functions and ecosystem services (adapted from Brussaard, 2012).

2.2.2. Impacts of land use/cove changes on soil quality

LULC change is the foremost cause for soil quality (SQ) change and information on the responses of SQ indicators is essential to develop a sustainable land management plan (Ayalew, 2015). Soil quality assessment with respect to land use change is very crucial particularly in tropical regions to improve land use systems (Yimer et al., 2006). The conversion of forest land into other land use systems such as cultivated land (CL) and grassland (GL) caused a decline in soil quality, thereby reducing its potential for actual productivity (Wei et al., 2014). Land use/land cover change in Sub-Saharan African region has brought the failure of soils to provide essential ecosystem services as a result SQ decline has become a major challenge (Diao et al., 2010).

LULC changes coupled with climate change are known to have causing soil degradation due to alterations in the individual soil quality parameter. For instance, the loss of SOC and soil microbial biodiversity is the result of the conversion of forest land into cultivated land which is widely documented in (Padbhushan et al., 2022; Nath et al., 2018). In addition, LULC change affects the distribution and supply of soil nutrients by directly altering soil properties and by influencing biological activities. Deforestation increases CO_2 and other GHG emissions contributing to climate change (Sanderman et al., 2017). Therefore, enhancing SOC sequestration plays a great role in reducing loss of SOC, mitigating climate change and reducing associated impacts of ecosystem services.

However, anthropogenic activities are the major proximity causes for environmental changes at various spatial scales (Hegazy and Kaloop, 2015). Soil quality deterioration and other environmental problems like soil erosion, climate change, loss of biodiversity, carbon cycle and impairment of ecosystem services are the consequences of land use/land cover changes (Lepcha and Devi, 2020; Wang et al ., 2016). In Africa, soil quality deterioration has been aggravated due to population pressure and subsistence farming associated with poor land use systems; in turn resulted in declining of agricultural productivity and food insecurity (Alarima et al., 2020; Obalum et al., 2012). Previous studies conducted in the highlands of Ethiopia indicated that land use land cover changes have many implications on soil degradation, soil quality deterioration and suitability of an area for specific use (Pham et al., 2018). Changes from natural forest to agricultural land are the major causes of extensive soil quality deterioration, depletion of soil nutrients and declining of agricultural production in the highlands of Ethiopia (Molla et al., 2022; Aredehey et al., 2019), Dagnachew et al. (2019) and Teferi et al. (2016) also showed that human-induced activities in the Upper Blue Nile Basin had a substantial impact on major soil quality indicators. The highlands of Ethiopia faced critical soil related problems from the perspectives of soil quality and fertility. These include: soil acidity that covers about 40% of cultivated fields; high rate of soil organic matter loss and depletion of macro nutrient (high rate of negative nutrient balances), Addressing the problems of soil fertility component has become a major policy issue in Ethiopia, since the agricultural sector is the major pillar of economic and social development (Neglo et al., 2021). Therefore, improving and maintaining soil quality is a pre-requisite for agricultural sustainability**.** Hence, soil quality evaluation is crucial to understand the basic problems of soil quality indicators; to design site-specific and effective management strategies (Rahman et al., 2020).

In Africa, agricultural lands, including marginal lands are intensively cultivated and become under pressure due to ever increasing population. In developing countries were the livelihoods of the community is dependent on agricultural activities, loss of soil fertility is the major constraint to food security and rural poverty (seifu et al., 2020). In these countries, soil nutrient losses from agricultural lands and their negative consequences are the highest in the world. Agricultural productivity is affected by fragmented ecosystems, low inherent fertility of soil and insufficient application of external inputs (Juilo and Carlos, 2006). Moreover, the application/utilization of appropriate soil management strategies still continues as a major problem in these countries (Onduru and Du Preez, 2007). In these regions, soil nutrient losses from agricultural lands and their negative consequences are the highest in the world. Hengl et al. (2015) reported that 80% of African agricultural land has soil fertility below threshold level as a result of progressive nutrient loss due to poor soil management systems. Even when compared to other African countries, agricultural sustainability is becoming a major threat in SSA due to soil erosion, low application of external inputs and low inherent soil fertility which are the major factors for soil nutrient depletion (Harris and Consulting, 2014; Van Beek et al., 2016). Therefore, the application of sufficient amount of external input is required to reverse progressively increasing depletion of macronutrients and to increase agricultural production in these countries (Zhang et al., 2020).

2.3. Soil erosion and sediment export

Soil degradation in the form of water erosion has become an important global concern because of its impacts on agricultural productivity, and food security as it increases the cost of production (Teferi et al., 2016; Ferreira et al., 2015). Soil erosion is a particular concern in Asia and Africa where agriculture is the bases for the livelihoods of the population (Tamene and Le, 2015). In the African continent about half of the population is affected by soil erosion which is the highest in the world (Blanco-Canqui and Lal, 2008). Soil erosion is particularly severe in agricultural land use systems causing the loss of nutrient-rich top soil which leads to a decline in agricultural productivity (Erkossa et al., 2015; Haregeweyn et al., 2017)

Ethiopia is one of the African countries that are severely affected by soil erosion (Samuel et al., 2016). Soil erosion is reportedly severe in the highlands because of the rugged terrain, inappropriate land management practices and socioeconomic characteristics of these regions. Land conversion to expand arable land often included the cultivation of marginal lands with steep slopes where the poor farmers cannot afford to construct terraces. In addition to this, the problem of over grazing and complete removal of crop residues from agricultural fields that predispose the soil to the impacts of the monsoon rains that result in massive soil erosion

(Erkossa et al., 2015; Addis et al., 2016). Other studies conducted in different parts of the country also confirmed this condition.

2.3.1. The rate of soil erosion in Ethiopia

Soil erosion rates aggregated for all land use types ranged between 3.4 and 85 t/ha/year depending on climate and land use type; the rates for cultivated fields was from 50 to 179 t/ha/year (Hurni et al. 2015; Adimassu et al. 2012). The results of Sonneveld et al. (2011) demonstrated the outputs of mean soil loss map at national level using different models. The soil erosion rate varied significantly and reaches to a maximum of more than 100t/ha/year in the north western part of the country. The soil conservation research project (SCRP) found the highest erosion rates of 110 t/ha/year at Anjeni experimental site in the Upper Blue Nile Basin (SCRP, 1996). Taye et al. (2013) also documented the significant impacts of land use and land cover changes on soil loss; with a soil loss rate of 38.7 t/ha/year from rangeland as compared to cropland (7.2 t/ha/year). Variations in land use (i.e., cover and management factors of erosion) and climatic conditions particularly rainfall (i.e., the R-factor) was found to explain about 35% of the differences in soil loss across different locations in Ethiopia. The findings of (Nyssen et al., 2009) from northern Ethiopian and from the Central Rift Valley region (Meshesha et al., 2014) reported larger erosive power of rainfall as compared to other areas in the country. Fenta et al (2021) demonstrated that soil loss was highly variable in Ethiopian River Basins with a national mean value of 16.4t/ha/yr. From this study it was observed that the highest soil loss rate was from Tekeze Basin (43.5 t/ha/yr) followed by Abay Basin (32.8 t/ha/yr) and Omo-Gibe Basin (22.1 t/ha/yr). Haregeweyn et al. (2017) also demonstrated the severity classes of soil erosion in various landscape positions of the highlands of Ethiopia (Table 2.1).

Table 2.1. Soil erosion severity classes in Ethiopia

Soil erosion rate (t/ha/yr)	Severity class	Reference
0-5	very slight	
5-15	slight	Haregeweyn et al. (2017)
15-30	Moderate	
30-50	Severe	
>50	Very severe	

Hurni et al. (1993) reported that soil loss tolerance level in Ethiopia ranged from 2-18 t/ha/yr, Soil loss assessment results of Ethiopian highlands revealed that soil erosion rates are greater than this soil loss tolerance level.

2.3.2. Impacts of soil erosion

Soil erosion causes adverse economic and environmental impacts which include soil quality deterioration, declining in agricultural productivity and sediment deposition overlying fertile topsoil, infrastructure destruction, siltation of dams and water reservoirs and damage to the irrigation canals (Tsegaye, 2019). FAO (2015) estimated that each year about 100,000 km^2 of crop land has been lost due to soil erosion. Soil erosion also affects other components of land resources like vegetation, water and air (Teferi et al., 2016).

The impacts of erosion are categorized as on-site and off-site or downstream effects.

2.3.2.1 On-site effects

Soil erosion is a dominant land degradation process that significantly degrades soil quality and soil fertility thus reducing agricultural productivity (Mekonnen et al. 2017; Tsegaye 2019). Soil nutrient losses and declining of agricultural productivity in the Ethiopian highlands are due to extensive soil erosion (Haregeweyn et al., 2008). The on-site effects are associated with declining crop yields as a result of loss of nutrients, water holding capacity and rooting depth of the eroding soils. The findings of Tamene and Vlek (2008) showed that soil erosion results in a 2.2% annual decline in land productivity and projected a 30% reduction in the per-capita income of the population of Ethiopian highlands. Soil fertility decline is manifested in terms of loss of nutrients (N, P, K) and organic matter-rich upper layers of the soil. With loss of organic matter-rich water holding capacity as erosion results in the breakdown of soil aggregates and the removal of smaller particles. In areas with severe soil erosion, the removed nutrients with sediments are three times higher than nutrients in the eroding soil (Narendra et al., 2017). The estimated proportions of top soil nutrients removed ranged from 1 kg to 6 kg for nitrogen, 1 to 3 kg for phosphorus and 2 to 30 kg for potassium. The on-site crop yield loss has been aggravated by soil nutrient depletion, arising from continuous cropping coupled with the removal of crop residues, low external inputs and the absence of adequate soil nutrient saving and recycling technologies. Gebreselassie et al. (2016) also explained the impact of soil erosion on agricultural productivity and its associated economic loss with an annual cost of US$4.3 billion.

2.3.2.2 Off-site impacts of soil erosion

Besides on-site effects, soil erosion also inflicts off-site impacts as a result of the movement of sediment and agricultural pollutants into the river courses (Setegn et al., 2009). In some cases, increased downstream flooding may also occur due to the reduced capacity of eroded soil to absorb water in the upper catchments. Sediment loads from unmanaged upland areas can lead to the silting-up of dams and water reservoirs, disruption of the ecosystems of lakes, and contamination of drinking water. Power generation reservoirs like Koka, Gilgel Gibe I, Aba Samuel, and Melka Wakena; water supply reservoirs including Angereb, Legedadi, Borkena and Adrako and irrigation reservoirs in the northern highlands (Wolka et al., 2015) are highly affected by sediment loads. For instance, the storage volume of Koka dam has been reduced due to sediment accumulation with the loading of 3.5 million m^3 of sediment in 23 years period (Zemadim *et al*., 2014). The results of Degfie et al. (2021) from Lake Hawassa catchment confirmed that the area of Lake Hawassa has increased by 232 ha in 16 years (on average, 14.5 ha/yr) as a result of sediment load (14,168t/yr) from the upland areas. The study of Yitaferu (2007) from Lake Tana Basin indicated that the capacity of the Lake Tana has decreased because of siltation with sediment loading rate of 14.84 million tons per year.

The loss of Cheleleka wetland in the Central Rift Valley of Ethiopia (Degife et al., 2019) was as the results of excessive soil erosion and subsequent siltation. Devi et al. (2008) also emphasized that siltation and nutrient enrichment are the main problems of Gilgel Gibe-I hydro-power dam. Mekonnen et al. (2022) also demonstrated that the capacity of Adebra NSR reservoir was decreased from 36,902 m^3 in 2012 to 27,722 m^3 in 2020 due to sediment load. These findings confirmed the loss of 24.8% its capacity during 8 years period. The average sediment deposition rate was estimated to be 1147.5 m3 /year. Abijatta Lake in the Central Rift Valley region of Ethiopia may dry out in the future because of siltation (Temesgen et al., 2014a). Similarly, in the East African region, the Sinnar, the Rosieres and the Khashmel Girba reservoirs in Sudan (Bekele et al., 2008) and the High Aswan reservoir in Egypt (Ahmed and Ismail, 2008) have lost substantial proportions of their planned storage capacities due to sedimentation.

Soil erosion also causes negative impacts on water quality and aquatic ecosystems due to the high concentration of sediments and associated nutrients from land into water bodies. The effects of sediment load on the performance of a water treatment plant include: high turbidity, high cost of coagulants to be used in water treatment, low quantity of water to be supplied, and fills of tanks and pipes with mud (Munyaneza, 2015). Most of the aquatic habitats are destroyed by eroded soil from the upland areas. The eroded soil flows in to rivers, lake and water reservoirs with a lot of quantity of chemicals and heavy metals which causes excess turbidity and water eutrophication that harms aquatic life and makes the water less useful for domestic use and recreation (Bing et al., 2013; Wilson et al., 2008).

2.4. Soil nutrient flows and balances at different spatial scales

2.4.1. Concepts of soil nutrient flows, stocks and balances

Soil nutrient flow is understood as the amount of soil nutrients that flow in and out of a system or area. Nutrient flow can be measured at different spatial scales, plots, farm, catchments, districts, regions, national, and higher levels.

Nutrient balance is the difference between the sum of nutrient input flows and the sum of nutrient output flows in agro-ecosystems with predefined spatial-temporal boundaries (Bindraban et al., 2000). This parameter helps us to explore the consequences of farming system on soil fertility as well as the sustainability of agroecosystems.

Soil nutrient stock refers to the total amount of nutrients found in the upper 30 cm of the soil depth. Both the nutrients in the organic matter fraction and nutrients absorbed in to the solid phase are considered part of the stock.

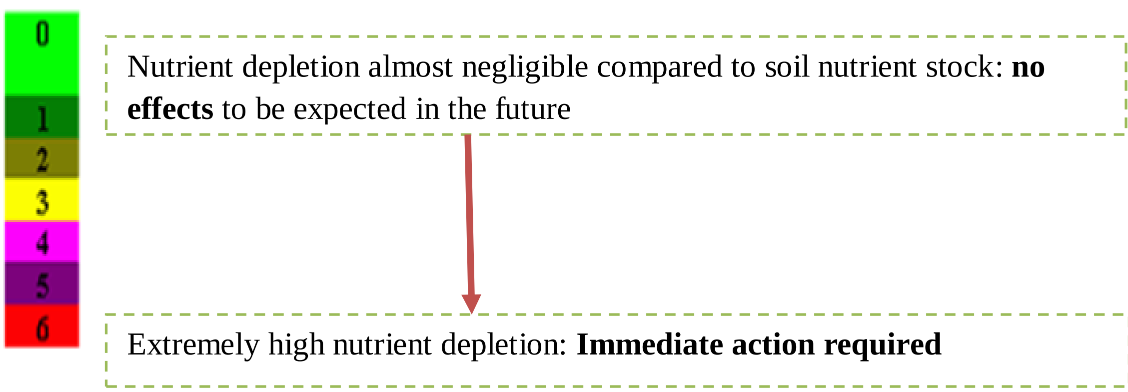
In Africa, the fertility level of soils is significantly variable and soils respond differently to external inputs. Low inherent soil fertility, limited replenishment of removed nutrients and high erosion rates in mountainous areas cause soil fertility decline that become a major threat to current and future food security (AGRA, 2014). Maintaining and improving soil fertility in this continent is crucial to attaining the Millennium Development Goals, especially contributing to 'Eradicating extreme poverty and hunger (MDG 1)' and 'Ensuring environmental sustainability (MDG 7)'. Despite major efforts from research centers, NGOs, and governments, applying effective soil fertility management strategies is still remain a major challenge in SSA (Onduru and Du Preez, 2007) as human-induced activities and biophysical factors are affecting the potential of soil resources in these areas. The main factors contributing to soil nutrient depletion through physical and chemical properties of soils together with climatic conditions in tropical Africa and particularly SSA, are generally loss of nitrogen (N) and phosphorus (P) through wind and water erosion; as well as leaching away of nitrogen and potassium (Henao and Baanante, 2006; Amede, 2003).Therefore, using soil quality indicators which are reliable to estimate the level of soil nutrient mining is crucial to identify agroecologies and farming systems with high level of soil depletion (Sheldrick and Lingard, 2004). According to Hartemink (2010) soil fertility decline can be assessed via expert knowledge systems, the monitoring of soil chemical properties over time or at different sites, and the calculation of nutrient balances, with the last one being the most used and cost-efficient technique.

In Africa, the nutrient balance approach became relevant starting from the study of Stoorvogel and Smaling (1990), and the research is still on the agenda (Vitousek et al., 2009). The average nutrient depletion rates for macro nutrients at continental scale were 20 kg N/ha/yr, 10 kg P/ha/yr and 20 kg K/ha/yr; whereas the depletion rates of these nutrients in East Africa (typical examples are Kenya and Ethiopia, where there is high soil erosion rate) were two times higher than the continental mean values (40 kg N/ha/yr, 20 kg P/ha/yr and 40 kg K/ha/yr). The study results in this continent showed evidence of widespread nutrient mining leading to severe nutrient deficiencies across ecological zones. High spatial variability of soil nutrient mining in the sub-humid and humid West and East Africa countries is due to the variation in the management of agricultural lands (Henao and Baanante, 2006). Soil nutrient depletions have been considered a serious threat to agricultural productivity and have been identified as a major cause of yield reduction in SSA (Smaling et al. 2013; Henao and Baanante, 2006). Moreover, nutrient stocks are not static and continued nutrient mining of soils would result in increased poverty, food insecurity, and social and political instability.

The findings of Elias (2004) and Elias et al. (1998) from southern Ethiopia also emphasized that the depletion of soil nutrients from cultivated fields is the most challenging problem to insure agricultural sustainability in Ethiopia. By the same author, it was confirmed that soil fertility management strategies are significantly variable across locations and monitoring the levels of soil fertility for agricultural lands helps to identify appropriate management strategies inorder to counter balance the depletion of soil nutrients. Moreover, quantification of soil nutrient flows (nutrients entering in to the system and leaving the system) are a good indicators of whether the farming system is sustainable or not. Nutrient flows and balances are not meaningful without the knowledge of nutrient stocks. Linking nutrient balances and flows to soil nutrient stocks creates a valuable indicator for sustainability assessment in agricultural land use systems (Bahr et al., 2015). Hence, the nutrient balance should be related to the nutrient stock to determine the sustainability of a system as described in (Table 2.2).

Table 2.2. Relationship between soil nutrient depletion and soil nutrient stock

Nutrient depletion	Soil nutrient stock				
	Very small	small	moderate	Large	Very large
Very low	3	2	1	0	0
Low	4	3	2	1	0
moderate	5	4	3	2	1
High	6	5	4	3	2
Very high	6	6	5	4	3



2.4.2. Soil nutrient flows and balances in Ethiopia

Ethiopia is among the sub-Saharan African (SSA) countries with very high nutrient depletion rates because of its mountainous topography and intensive farming systems (Elias et al., 2019; Teferi et al., 2016). More soil nutrients are exported compared to natural and anthropogenic inputs (Van Beek et al., 2016; Haileslassie et al., 2005). The presence of high nutrient mining is an indicator of poor management of the soil resource. Nutrient balance results from previous studies, irrespective of the type of balances, spatial scale, and units, indicated that most systems had negative N and K balances. These values become more negative in regions where land users are extensively mining soil resources for their livelihoods with less/no supply of external inputs to their farmland. For P the trend was less remarkable. Based on the findings of Van Beek et al. (2016), the national average soil nutrient balances were -32, 9 and -7 kg/ha/yr for N, P and K respectively (Table 2.3). Negative P balances were found in some locations and most locations showed positive values. The variation between districts/sites might be attributed to the difference in the agroecology, land use type and management strategies. The study of Haileslassie et al. (2005) also showed that the national averages of nutrient balances were estimated at -84 kg N, -4 kg P and -53 kg K/ha/year, which is among the highest nutrient depletion rates for sub-Saharan Africa. A negative nutrient balance indicates that more nutrients are exported from the system than supplied into the system. The most contributors to nutrient exports are soil erosion and harvested products (Haileslassie et al., 2006). Van Beek et al. (2016) also showed that soil erosion accounted for about 50 % of the nutrient losses at a national scale.

Table 2.3. Nutrient balances in different spatial scales in Ethiopia (kg/ha/yr)

Study area	Spatial scale	N	P	K	Reference
CASCAPE project sites of the country	District	-32	9	-7	Van Beek et al. (2016)
Northern Ethiopia	Catchment	-41	1	-36	Gebremedhin et al. (2014)
Southern Ethiopia	Farm	+68	+7	-23	Haileslassie et al. (2006)
Central Ethiopia	Farm	-50	-4	-64	Haileslassie et al. (2006)
Western Ethiopia	Catchment	-46	+3	-75	Haileslassie et al. (2006)
Ethiopia	national	-122	-13	-82	Haileslassie et al. (2005)
Tigray Region	Catchment	-65	-6	-34	Assefa et al. (2005)
Southern Ethiopia	National	-92	+5	-49	Elias (1998)
Ethiopia	National	-47	-7	-32	Stoorvogel and Smaling (1990)
Western Ethiopia	Farm	+3	+5	-	Aticho et al. (2011)
Northern Ethiopia	Farm	-20.9 to -61.4	-0.7 to +11	-26.7 to -37.8	Gezie (2019)
Northern Ethiopia	Farm	-26.2 to -17.9	+3.9 to +6.7	-5.2 to -+2.9	Mesfin et al. (2021)
Northern Ethiopia	Catchment	-21 to -17.7	+3.5 to +8.0	-12.8 to -5.6	Kiros et al. (2014)
North western Ethiopia	Catchment	−112.4 to −89.4	−23.6 to -20.6	-130.2 to −124.7	Lewoyehu et al. (2020)

2.5. Over view of the methodological approaches used in this study

2.5.1. Application of GIS and remote sensing in LU/LC and LST change analysis

Using remote sensing (RS) data together with Geographic Information Systems (GIS) proved the effectiveness of this technique for monitoring the spatio-temporal dynamics of land use/cover (LULC) and land surface temperature (LST) (Kimuku and Ngigi, 2017). The use of satellite images and data to assess and analyze changes in land use and land cover (Chowdhury et al., 2020; Congedo, 2021) has become the most recognized and powerful technique to obtain more accurate information on land surface characteristics at different temporal and spatial scales. Therefore, RS can be used to carry out studies at different spatial and temporal scales (Haylemariam, 2018) in such a way that information can be useful for monitoring changes in LULC and LST and the relationship between these parameters can be established.

2.5.1.1. Methods to Retrieve LST from remotely sensed data

Nowadays, one of the principal methods replacing the use of information obtained from surface stations is the retrieval of LST through RS techniques. It has been demonstrated to be one of the preferred methods to analyze LST because a product with better spatial coverage can be obtained in near real-time (Schuch et al., 2017). For this purpose, several algorithms can be used to calculate LST, including the use of the single-channel algorithm to retrieve the LST of products derived from Landsat satellites (Mujabar, 2019; Haylemariyam, 2018).

A single channel algorithm could be applied following the process described by Mujabar (2019). The first step to retrieve LST is the conversion of digital number (DN) to radiance (Lλ) as described in equation 1(for Landsat 5 and 7). The thermal band with spectral radiance values would be converted to effective at sensor brightness temperature (T_B,). Brightness temperature includes atmospheric effects such as absorption and emissions. This parameter considers the Earth’s surface as a black body (Chander et al., 2009).

$$L\lambda = \frac{(\mathbf{LMAX\lambda - LMIN\lambda})}{(\mathbf{QCALMAX - QCALMIN})} * (\mathbf{QCAL - QCALMIN}) + \mathbf{LMIN\lambda}$$

Where, Lλ= at sensorspectral radiance; QCALMAX = maximum DN value of pixels; QCALMIN = minimum DN value of pixels; QCAL = DN value of the pixel; LMAXλ = maximum spectral radiance of the sensor and LMINλ = minimum spectral radiance of the sensor

$$T_B = \left(\frac{\mathbf{K2}}{\mathbf{ln}\left[\left(\frac{\mathbf{K1}}{\mathbf{L\lambda}}\right) + \mathbf{1}\right]} \right)$$

Where T_B is effective at-sensor brightness temperature (°K); band 6 calibration constants (W/m2 srµm) are K1= 607.76 and K2 = 1260.56, whereas band 10 calibration constants are K1= 774.89 and K2 = 1321.08.

Once TB is calculated, it is necessary to estimate the land surface emissivity, which is a crucial factor for LST retrieval. Land surface emissivity was calculated using the following equation (Mujabar, 2019) :

(Ɛ) = 0.004*PV + 0.986

Where, Ɛ is the land surface emissivity (dimensionless); PV is the Proportion of Vegetation (dimensionless). PV can be calculated using equation 6, which involves the use of the Normalized Difference Vegetation Index (NDVI).

$$\boldsymbol{PV} = \left(\frac{\textbf{NDVI}-\textbf{NDVI min}}{\textbf{NDVI max}-\textbf{NDVI min}}\right)^{2}$$

$$\textbf{NDVI} = \left(\frac{\boldsymbol{Band\ 5}-\boldsymbol{Band\ 4}}{\boldsymbol{Band\ 5}+\boldsymbol{Band\ 4}}\right)$$

Where NDVI (dimensionless, values from -1 to 1) can be obtained through equation 5 reported by Van de Griend & Owe (1993). Finally, LST was calculated using the following equation Mujabar (2019).

$$\textbf{LST} = \frac{\text{BT}}{\left[\!\left[1+\left\{\left(\text{L}\lambda * \frac{\text{BT}}{\rho}\right) * \ln \varepsilon\right\}\right]\!\right]}$$

Where, T_B corresponds to effective at sensor brightness temperature (°K); Lλ is the central wavelength of the emitted radiance; ρ is a constant equal to 14,380 µmK (ρ =h × c/σ, where σ is the Boltzmann constant =1.38×10−23 J/K, h is the Planck's constant = 6.626×10−34 Js, and c is the velocity of light = 2.998×1014 µm/s); Ɛ is the land surface emissivity.

All these analyses can be done using GIS and remote sensing. Hence, this technique is a power full tool for retrieving remotely sensed data and analyzing land use/cover and land surface temperature changes.

2.5.2. Integration of GIS with soil erosion models in quantifying soil erosion and mapping its spatial distribution

Understanding the extent and magnitude of soil erosion risk is a prerequisite to design effective management strategies (Benavidez et al., 2018). soil erosion assessment methods can be grouped into three main approaches: a runoff plot experiment that provides net soil loss (Hurni, 1985; Herweg and Stillhardt, 1999), a field survey that involves the measurement of visible soil erosion indicators and the combination of erosion-influencing factors (Whitlow, 1986), and erosion modelling that involves the use of empirically derived equations or process-based models (Wischmeier and Smith, 1978; Helldén, 1987). Soil erosion models are the most commonly used options to assess soil erosion risk, sediment flux, and design and evaluation of the effectiveness of soil conservation and management technologies (Easton et al., 2008). Different erosion and sediment models exist focusing on different spatial and temporal scales with different degrees of complexity and precision to address the practical implication of soil erosion at the landscape level. However, researchers (Coppus and Imeson, 2002) proved that there is no single erosion model that can be universally accepted and applied. There is also no clear agreement in the scientific community that which kind of model is more appropriate to estimate soil erosion in a given catchment since those models have their own potentials and limitations (Tamene et al., 2006).

Based on the nature of the basic algorithms, there are three main types of erosion models. According to Wheater et al. (1993) and Argent et al. (2005), these models are classified as empirical, conceptual and physical based models. Among these models, empirical models are simple and widely applied in data scarce regions. They are based on extensive experimental results (site-specific observations) and input-output relationships. The data and computational requirements for such models are usually less than those from conceptual and physically-based models (Li et al., 2009). These models have limitations of applicability in areas where data were not used for model development (Merritt et al., 2003) but such models simply calibrate a relationship between inputs and outputs without any effort to describe the condition caused by each processes (Argent et al., 2005). Typical examples of this model include the Universal Soil Loss Equation (USLE), Modified Universal Soil Loss Equation (MUSLE) and Revised Universal Soil Loss Equation (RUSLE). Conceptual models usually represent catchments as a series of internal storages (Chandramohan et al., 2015). They define the general mechanisms that govern the interchange of sediment and water between these storages. Input parameters are usually obtained in calibration with respect to field-measured data, and evidence shows that determining the optimal set of values can be cumbersome and in some cases even harder to determine (Merritt et al., 2003).

Physical (process)-based models are based on the understanding of the physics of flow and sediment transport processes and their interaction using equations governing the transfer of mass, momentum and energy (Kandel et al., 2004). These models are based on the computation of erosion using mathematical representations by knowing the concepts and main processes of hydrology and soil erosion (Argent et al., 2005). They are commonly applied to small catchments represented by detailed data. Examples of this model are: Water Erosion Prediction Project (WEPP); European Soil Erosion Model (EUROSEM) (Morgan et al., 1996). The main limitation of these models is the requirement of intensive data for model parameterization, and calibration and more particularly the lack of data for validating the spatial pattern of runoff, sediment and soil nutrients losses in order to apply a model to a wide range of field conditions. The other major limitation of these models is that they are too complex and their reliance on data to test and calibrate for assessment of performance before the model output is used for decision making (Argent et al., 2005). It is difficult to reliably apply most of the physical-based models developed in the data-rich regions to developing countries, where both data availability and quality are critically poor. Selection of appropriate model(s) that can suit the areas under study is therefore crucial and needs to be based on the objective, availability of data and other resources and scale of investigation required.

2.5.3. Soil quality assessment methods

The ultimate purpose of assessing soil quality is to improve agricultural productivity, water quality, and habitats of all organisms including human beings (USDA, 2006). For soil quality assessment, integration of static and dynamic soil chemical, physical, and biological quality indicators need to be defined in order to identify different managements and environmental scenarios. Several methods of soil quality evaluation have been developed, including soil card design and test kits, geostatistical methods or soil quality index methods (Bünemann*et al.*, 2018**)**. In general, soil quality assessment is carried out by selecting a set of soil properties which are considered to be indicators of soil quality (Vasu *et al.,* 2016). The use of one indicator or indices integrating only two parameters has many limitations and provides insufficient information about soil quality and degradation (Bastida*et al.,* 2008; Masto *et al.,* 2015). Recently, the new methodological approach (soil management assessment frame work) that focused on the derivation of multi-parametric indices by combining different parameters (soil physical, chemical and biological quality indicators) has been widely used. This approach involves three steps: indicator selection, indicator interpretation, and integration into a soil quality index (Andrews et al., 2004). A soil quality index (SQI) could be defined as a minimum set of parameters that provides numerical data concerning the capacity of soil to carry out one or more functions (Garrigues *et al.,* 2012; Asensio *et al.,* 2013). The selection of a minimum soil data set (MDS) is based on either expert opinion (subjective), or mathematical and statistical (objective) methods (Bastida *et al.,* 2008; Bünemann *et al.,* 2018). In recent times, statistical data reduction by using multivariate techniques such as principal component analysis (PCA), redundancy analysis, discriminate analysis and multiple regressions have become more common.

2.5.4. Methodological approaches and scale issues in soil nutrient balance studies

Soil nutrient balance studies conducted in Africa adopted the method of Stoorvogel and Smaling (1990), which accounted for five nutrient inflows and five nutrient outflows. For those fluxes which are difficult to measure, transfer functions/regression equations were integrated to quantify nutrient balances (Bindraban et al., 2000; Lesschen et al., 2007). Nutrient balances can be assessed using partial and full nutrient balance approaches (Haileslassie et al., 2005). The partial nutrient balance approach considers easily measured parameters, which include two nutrient inflows (organic and inorganic inputs) and two outflows (harvested crop yield and residue) (FAO, 2003). The numerical value of nutrient balance for macro nutrients (N, P and K) for a given agroecosystem is the difference between the sum of nutrient inputs and nutrient outputs. The spatial scale of the study of nutrient balance ranged from plot level to continental level depending on the objectives of the study. The scales include: Macro-scale, Meso-scale, and Micro-scale (FAO, 2003). Macro-scale studies were undertaken at a country and continental level using the methodological approach of Stoorvogel and Smaling (1990). Soil nutrient balance studies conducted in Sub-Saharan Africa by FAO (2003) are typical examples of this. Meso-scale studies reflect soil nutrient balance analysis at the district level. Data inventories and identification of soil management strategies are better than the macro-scale approach. Micro-scale studies considered farms/agroecosystems as the unit of soil nutrient balance studies. In this approach, the socio-economic status of household groups and integrated nutrient management aspects could be addressed. Moreover, nutrient balance studies at the micro-scale differ considerably from macro and meso scales as they are generally based on on-site inventories and monitoring studies that provide first hand data. However, they all utilize regression models to estimate difficult flows, such as leaching and gaseous losses.

CHAPTER THREE: MATERIALS AND METHODS

3.1. Description of the study area

3.1.1 Location and Biophysical characteristics

Suha watershed is geographically located between 37º 56' 15'' and 38º 18' 49''E and 10º 06' 46'' and 10º 41' 56''N in the north western highlands of Ethiopia and covers an area of 80,340 hectares (Fig 3.1). The altitude ranges from 1040 to 3986 meters above sea level (m.a.s.l) and characterized by complex topographic features that extend from flat to very steep slopes. Based on FAO (2006) slope classes, the watershed has seven slope categories (Table 3.1). Flat to very gently sloping (0-2% slope), gently sloping (2-5% slope), sloping (5-10% slope) strongly sloping (10-15% slope) and moderately steep (5-30% slope). Steep (30-60% slope) and very steep (>60% slope) classes characterize the upper part of the watershed.

Table 3.1. Slope classes and proportion of each class in Suha watershed (based on FAO, 2006 classification)

No.	Slope (%) range	Slope class	Area (ha)	Proportion (%)
1	0-2	Flat to very gently sloping	2586.7	3.2
2	2-5	Gently sloping	10748.4	13.4
3	5-10	Sloping	19234.7	23.9
4	10-15	Strongly sloping	12380.7	15.4
5	15-30	Moderately steep	19471.7	24.2
6	30-60	Steep	12231.2	15.2
7	> 60	Very steep	3686.9	4.6
	Total		80340.3	100

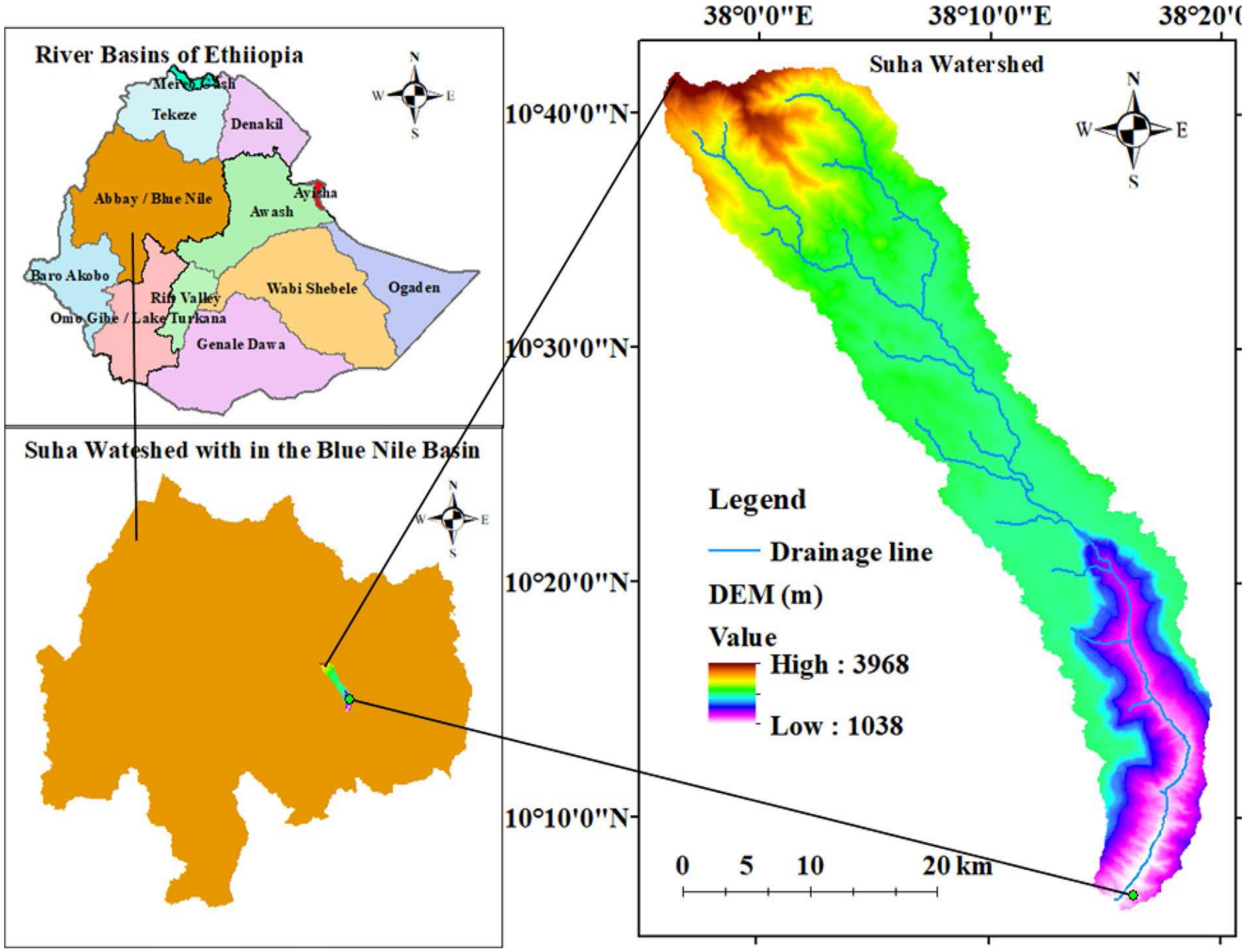


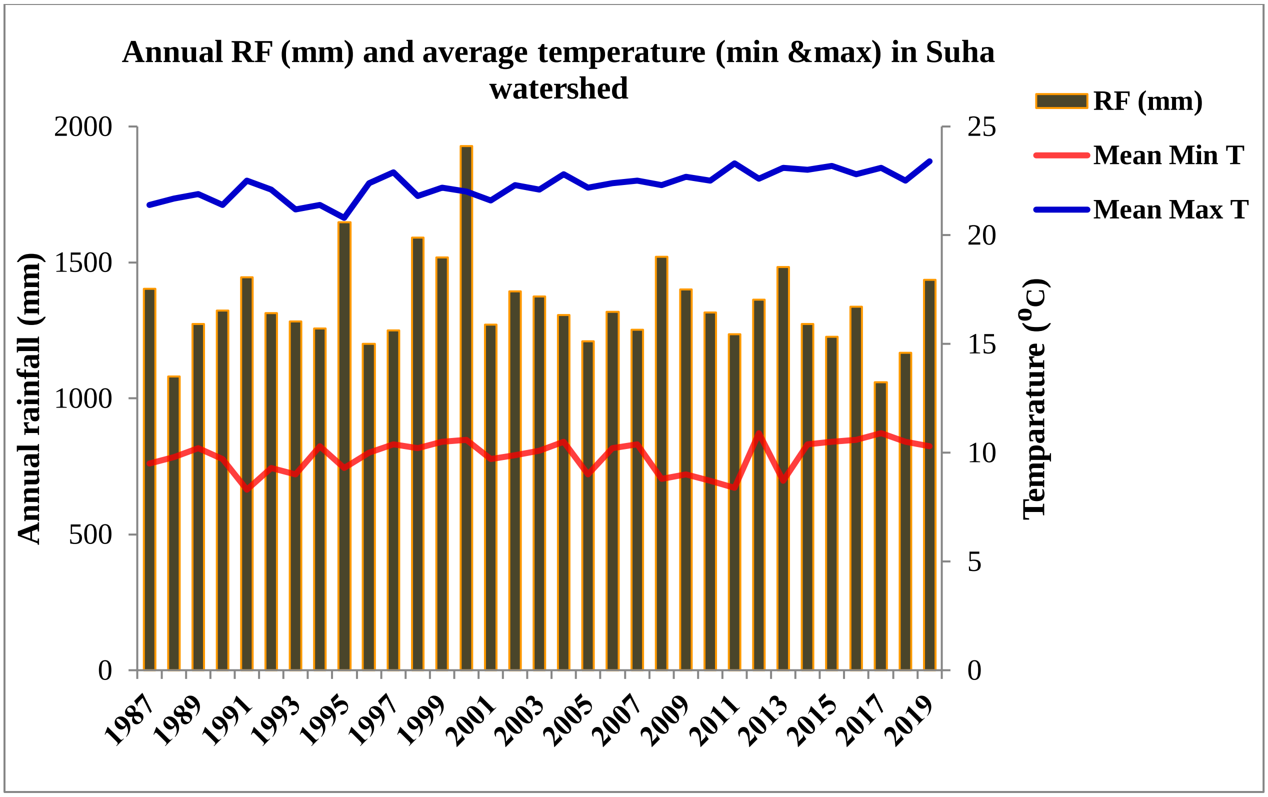
Fig 3.1. Location map of the study area

Based on the classification of Elias (2016) and MoARD (2005), the study area falls under the Tepid sub-humid mid highlands agro-ecological zone. In this classification, the basic parameters considered were climatic variables (temperature and moisture) and elevation. High rain fall coupled with poor land use systems in sloping areas is the cause of soil erosion and sediment source (Monsieurs et al., 2015). The climatic features are characterized by tepid mid highlands. Based on the long-term weather data obtained from nearby metrological stations (i.e., Bichena, Dejen, Kuy and Robgebeya), the area receives means annual rainfall in the range of 1213 mm at the lower part to 1396 mm in the upper part of the watershed (Fig 3.2). The mean minimum and mean maximum temperatures of the area are 8^{O}c and 25^{O}c. The rainfall pattern is characterized by a unimodal type with rains falling during the monsoon season (June to September) with a peak in August.

The geology of the Upper Blue Nile Basin is part of Shield Volcanoes and the Semen Mountains of the Ethiopian flood basalt plateau (Williams, 2016) dominated by alkali basalts, although metamorphic and sedimentary rocks can also be found in the lowlands of the basin (Ermias, 2015). According to the Abay Basin Master Plan (MoWR, 2011), the major soil types are Vertisol (57.3%), Leptosol (26.7%), Cambisol (9.6%), Luvisol (6.1%) and Nitosol (0.3%) were present; the central part of the watershed is predominantly covered by Vertisol. The distribution of soil types is mainly slope dependent.

Land use systems

Based on the Landsat image (2019) analysis results, six land use land cover classes were identified (Yeneneh et al., 2022). The largest proportion of land use class is cultivated field which accounts for 74 % of the catchment area. The other land use types include: grazing land (9.1%), forest land (1.3%), shrub land (4.9%), bare land (8.4%) and built-up area (2.0%). Currently *Eucalyptus* plantation is expanding in farm lands for different purposes. Agriculture is based on a mixed crop-livestock farming system. Major crops in the upper catchment include food oats (*Avena sativa*), barely (*Hordeum vulgare*), potato (*Solanum tuberosum*), bread wheat (*Trticum vulgare*) and onion (*Allium cepa*) are cultivated. In the middle part teff (*Eragrostis teff*) is predominantly cultivated and in the lower part sorghum is the dominant crops.



Figs 3.2. The long-term (1987 - 2019) mean annual rainfall and air temperature of the study area.

Source: National Meteorological Service Organization.

3.2 Land use and land cover change detection

3.2.1 Satellite image pre-processing

The general frame work of the methodology was depicted in (Fig 3.3). For this study various data inputs were collected from different sources. The main data sources were Landsat satellite images, digital elevation model (DEM) and referenced data. Landsat images for the periods of 1985 (TM), 1999 (TM), 2009 (ETM+) and 2019 (OLI_TIRS) were freely obtained from the United States Geological Survey (USGS) (http:// earthexplorer.usgs.gov). Satellite datasets were projected to the Universal Transverse Mercator (UTM), map projection system zone 37N and datum of World Geodetic System 84 (WGS84). To get better cloud-free images and avoid variation because of seasons dry season periods (January and February) were preferred for image downloading (Belay and Mengistu, 2019). In addition, ground truth data were gathered using Global Positioning System (GPS) during the field survey and additional reference datasets were generated from Google Earth for supervised classification, and accuracy assessment of classified images.

The first step in Landsat image analysis was preprocessing, which corrects distortions of satellite images and ready them for further analysis (Young et al., 2017; Mancino et al., 2014). Land sat sensors, solar radiation, atmosphere and topographic features are major factors for distortions of satellite images. Layer stacking, radiometric calibration, and atmospheric corrections were conducted using ENVI 5.3.software. Detailed descriptions of satellite images used for this study are given below (Table 3.2). The study watershed was extracted (prepared) using Arc Hydro tool in ArcGIS.

Supervised classification technique was employed in this study and a support vector machine (SVM) algorithm was applied to classify land use and land cover classes from Landsat5, Landsat 7 and Landsat 8 data. Different scholars showed that this algorithm has better accuracy compared with other algorithms, like maximum likelihood classification (Ahmad et al., 2018; Mondal et al., 2012). Support vector machine is different from other classification algorithms because of the way they choose the decision boundary that maximizes the distance from the nearest data points of all the classes. The supervised classification technique is recommended compared to unsupervised classification because of its quality result, provided that sufficient training data are available for the study area (Belay and Mengistu, 2019; Congalton, 2009). It is obvious that for supervised classification techniques reference data (ground truth data) are required. Hence, referenced data (ROIs for classification) were collected for different features using Google Earth for the study periods (1985, 1999, 2009 and 2019), Spectral separability of ROIs has been analyzed and pair separation results were compared based on Jeffries-Matusita, Transformed Divergence. Based on this classification technique, six land use/cover types were identified (Table 3.3).

Table 3.2. Details of landsat images used in this study.

Landsat image	Sensor_ID	Date of acquisition	Path _Row	No. of bands	Resolution (m)	Source
Landsat 5	TM	1985-01-09	169 & 53	7	30	USGS
Landsat 5	TM	1999-02-01	169 & 53	7	30	USGS
Landsat 7	ETM+	2009-02-04	169 & 53	8	30	USGS
Landsat 8	OLI_TIRS	2019-01-23	169 & 53	11	30	USGS

Table 3.3. Land use and land cover categories in suha watershed.

LULC category	Description
Agricultural land	Areas used for annual crops production which includes cereals, pulses, horticultural crops and oil crops
Grazing land	Areas covered with grass and used for communal grazing land
Forest land	Sites covered by dense forest with closed canopies (eucalypts plantation found in homesteads and farm lands also considered in this class).
Shrub land	Refers to small trees, bushes and shrubs with some grasses.
Bare land	Includes degraded areas, over-exploited cultivated farmlands, exposed bade rocks
Built up	Sites covered by buildings both in urban and rural areas, institutions and road infrastructures.

Accuracy assessment

This method is an important component of land use land cover classification and it is used to compare classified images with the ground truth data. On the basis of confusion matrix results, it is possible to evaluate whether an image is correctly classified or not. Referenced data sets (ROIs for validation) were collected for each period (1985, 1999, 2009 and 2019) using Google Earth Maps for each land use/ land cover class. A total of 300 reference data were collected using a random sampling technique for accuracy assessment. Over all accuracy and kappa coefficient values were analyzed by confusion matrix using the ground truth ROIs tool in ENVI software. Kappa coefficient result indicates overall agreement and its value lies between 0 and 1 (Foody and Mathur, 2004). User's accuracy and producer's accuracy were also analyzed to evaluate the accuracy of classified images of each land use/ land cover class. User's accuracy measures error of commission or false positive, whereas producer's accuracy measures error omission or false negative. Overall accuracy and Kappa coefficient were calculated using the following equations (Congalton, 2005).

$$\mathbf{OA = (X/Y) * 100} \qquad \text{Eq (1)}$$

Where, OA is the overall accuracy, X is the correct value in the diagonal of the matrix and Y is a total number of values taken as a reference point.

$$\mathbf{K} = \frac{\mathbf{N}\sum_{\mathbf{i=1}}^{\mathbf{r}} \mathbf{xii} - \sum_{\mathbf{i=1}}^{\mathbf{r}} (\mathbf{xi+} * \mathbf{x+i})}{\mathbf{N^2} - \sum_{\mathbf{i=1}}^{\mathbf{r}} (\mathbf{xi+} * \mathbf{x+i})} \quad \text{Eq (2)}$$

Where, K is the kappa coefficient, r is the number of rows in the matrix, xii is the number of observations in row i and column i, xi+ are the marginal totals of row i, x + i are the marginal totals of column i, and N is the total number of observations.

Change detection

Change detection is a post-classification technique that helps us to evaluate the conversion of one land use/land cover class to another category for multi-temporal classified images (Alawamy et al., 2020; Young et al., 2017). Change matrix analyses were employed for images of consecutive periods of 1985 and 1999, 1999 and 2009, 2009 and 2019, 1985 and 2019 using a change detection tool in ENVI 5.3 software. From the results of the change detection matrix, it is possible to present loss (error of omission) and gain (error of commission) for each LULC class for a given period of analysis. Percent of change and rate of change between periods were computed using the following equations (Temesgen et al. 2014a).

$$\text{Percent change} = \left(\frac{\text{Area } final - \text{Area } initial}{\text{Area } initial}\right)*100 \quad \text{Eq (3)}$$

$$\text{Rate of change (ha/yea)} = \frac{\text{Area } final - \text{Area } initial}{\text{Time interval}} \quad \text{Eq (4)}$$

3.2.2 Extraction of land Surface Temperature (LST) from Thermal bands

Thermal bands of Landsat images (Landsat 5, 7 and 8) were used to generate LST values for the study period (1985 - 2019). Band 6 and band 10 were used for Landsat5, Landsat 7 and Landsat 8 respectively. Thermal band 11 of Landsat 8 was not considered in this study following USGS recommendation (Avdan and Jovanovska, 2006). Mono-window algorithm used by other scholars (Imran et al., 2021; Kafy et al., 2020; Pal and Ziaul, 2017) was adopted for this study. The following steps were followed to generate LST from thermal bands.

The first task was the conversion of the digital number (DN) of the thermal band to the top of the atmosphere (TOA) radiance (Lλ). Satellite sensors measure reflectance from the earth's surface as digital numbers (DN) representing every pixel of the image. Because of this, the step is a prerequisite for the upcoming activities. For Landsat 5 and Landsat 7, the following equation was applied to extract spectral radiance from DN (Landsat 7 Data Users Handbook, 2019).

$$\mathbf{L\lambda} = \frac{(\mathbf{LMAX\lambda - LMIN\lambda})}{(\mathbf{QCALMAX - QCALMIN})} * \mathbf{(QCAL - QCALMIN) + LMIN\lambda} \qquad \text{Eq(5)}$$

Where, Lλ is at-sensor spectral radiance;

QCALMAX = maximum DN value of pixels;

QCALMIN = minimum DN value of pixels;

QCAL = DN value of pixel,

LMAXλ = maximum at-sensor spectral radiance and

LMINλ = minimum at-sensor spectral radiance;

For Landsat 8, band 10 was used to extract spectral radiance. Calibration notifications from USGS indicated that data obtained from the thermal infrared sensor (TIRS) Band 11 has uncertainty and suggested using TIRS Band 10 data as a single spectral band for LST estimation (Rongali et al. 2018). The following equation described by Chander and Markham (2003) was employed for conversion of the digital number (DN) to the top of the atmosphere (TOA) radiance (Lλ).

$$\mathbf{L\lambda = ML*QCAL + AL} \qquad \text{Eq (6)}$$

Where, Lλ = Spectral Radiance of top of the atmosphere;

ML = the band- specific multiplicative rescaling factor (0.0003342) (Table 3.4);

AL = the band- specific additive rescaling factor (0.1) and;

QCAL = Digital number (DN) of band 10

The next step was the conversion of spectral radiance to satellite brightness temperature (BT) as follows (equation 7).

$$\mathbf{BT} = \left(\frac{\mathbf{K2}}{\mathbf{ln}\left[\left(\frac{\mathbf{K1}}{\mathbf{L\lambda}}\right) + \mathbf{1}\right]}\right) - 273.15 \qquad \text{Eq (7)}$$

Where, BT = At-satellite brightness temperature;

Lλ = TOA spectral radiance (equation 6);

K1 and K2 are calibration constants (Table 3.5) and;

λ = spectral radiance in watts per meter squared steradian micron (W/ (m^2*sr*µm)).

The third step was calculating the proportion of vegetation (PV). Vegetation cover (greenness) is one factor for the spatial variation of land surface temperature. For this reason, input data used to estimate PV value was the Normalized Difference Vegetation Index (NDVI) which is calculated using equation (9) as proposed by Wang et al., (2015); PV was calculated using the method proposed by Sobrino and Romaguera (2004).

$$\boldsymbol{PV} = \left(\frac{\textbf{NDVI}-\textbf{NDVI min}}{\textbf{NDVI max}-\textbf{NDVI min}}\right)^{2} \qquad \text{Eq(8)}$$

$$\textbf{NDVI} = \left(\frac{\boldsymbol{Band\ 5}-\boldsymbol{Band\ 4}}{\boldsymbol{Band\ 5}+\boldsymbol{Band\ 4}}\right) \qquad \text{Eq(9)}$$

Where, band 5 is near infra-red (NIR) and band 4 is red

The fourth step involves the calculation of surface **e**missivity (E): It is an important parameter to estimate LST (Meng et al., 2019). The value of this variable is obtained by taking the ratio of two different emitted radiances (actual emitted radiance and radiance from the perfectly emitting surface) under the same temperature condition (Norman and Becker, 1995). Hence, accurate estimation of land surface emissivity is a pre-requisite to extract land surface temperature from Landsat images. It was calculated using the proportion of vegetation (PV). The following equation was applied to estimate this parameter ($\mathcal{E}$) (Wang et al., 2015).

Land surface emissivity ($\mathcal{E}$) = 0.004*PV + 0.986 Eq (10)

The last step was estimating land surface temperature (LST) using the following equation proposed by Artis & Carnahan (1982) and Jeevalakshmi et al. (2017).

$$\textbf{LST} = \frac{\text{BT}}{\left[\!\left[1+\left\{\left(\text{L}\lambda*\frac{\text{BT}}{\rho}\right)*\ln\varepsilon\right\}\right]\!\right]} \qquad \text{Eq (11)}$$

Where,

LST = Land surface temperature;

BT = at satellite brightness temperature (K);

Lλ = wave length of emitted radiance in meters;

ρ = h*c/ σ =emitted radiance: ($\rho = h * c/\sigma = 1.438 * 10- 2$ Mk);

σ = the Stefan–Boltzmann constant;

h = is Planck's constant;

c = the velocity of light, and

ε = the land surface emissivity (LSE).

Table 3.4. TM, ETM+ and TIRS thermal band calibration constants (K1 and K2 values)

Thermal bands	Constant1- K1 (watts/(meter squared × ster × µm))	Constant 2- K2 (Kelvin watts/(meter squared × ster × µm))
Landsat 5 (band 6)	607.76	1260.56
Landsat 7 (band 6)	666.09	1282.71
Landsat8 (band 10)	774.89	1321.08

Table 3.5. Rescaling factors for Landsat 8 (band 10)

Rescaling factor	Band 10	Source of data
M_L (Radiance mult. band)	0.0003342	Meta data
A_L (Radiance add band)	0.1	Meta data

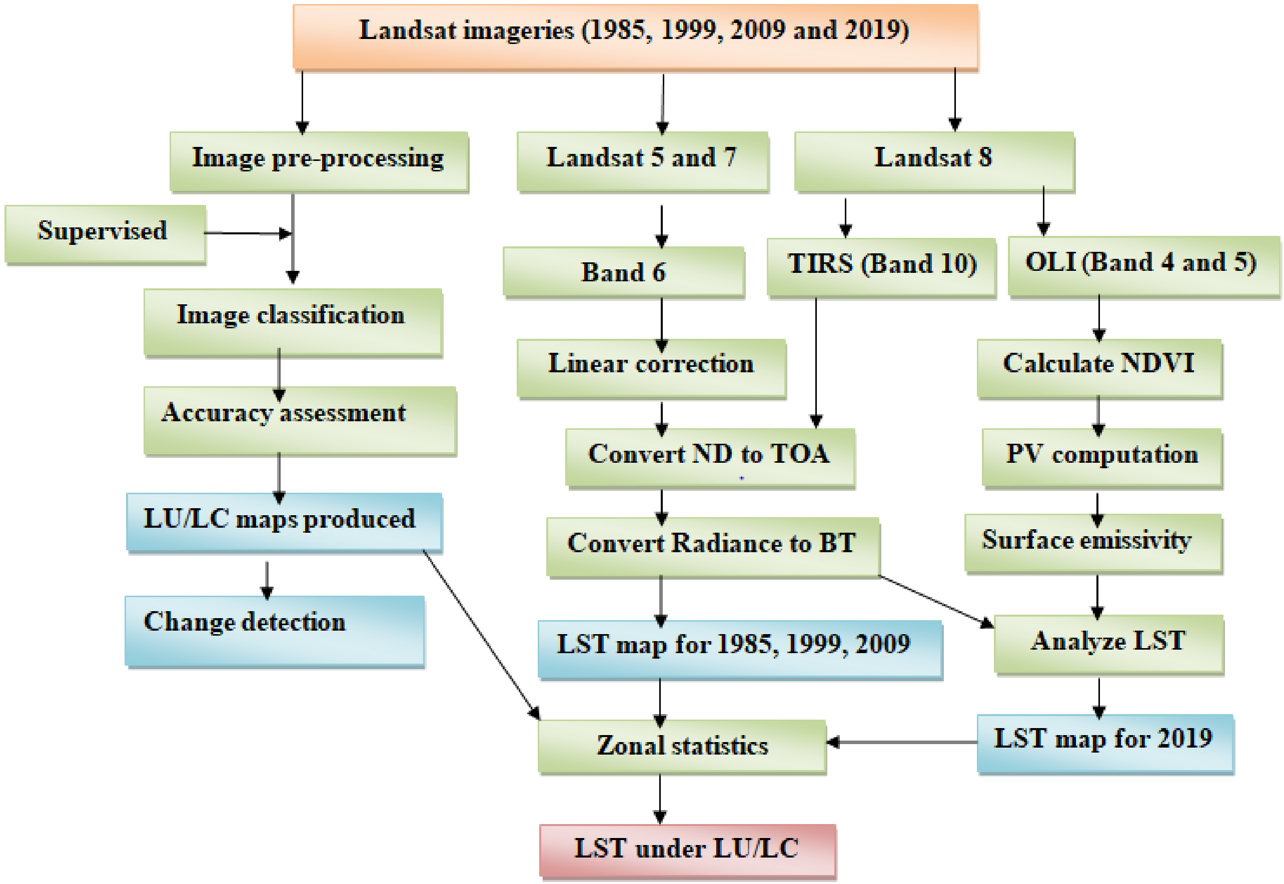


Fig 3.3. Methodological frame work applied for LULC and LST analysis.

3.3 . Quantification of soil erosion and sediment delivery

Soil loss and sediment yield were estimated using various data sources (Table 3.6). The Digital Elevation Model (DEM) of the study area derived from ASTER (Advanced Space borne Thermal Emission and Reflection Radiometer) with a spatial resolution of 30m * 30 m was used as data input to generate the slope map and to calculate the slope length and slope steepness factor (LS-factor) of the study area. Land cover factor (C_factor) and management/conservation practice (P_factor) were generated from land use/cover raster maps. The rainfall data was obtained from the National Meteorological Agency, Ethiopia and soil data from Abay Basin Master Plan (MoWR, 2011). These data inputs were integrated in RUSLE model using ArcGIS 10.5 software.

Table 3.6. Data inputs used for RUSLE model to estimate soil loss and sediment yield

Data type	Source	Reference
DEM	ASTER	USGS (http://gdex.cr.usgs.gov/gdex)
Soil	Abay Basin master plan project	MoWR, 1998 (Abay Basin master plan project)
Rain fall	Data from stations (1974 - 2019)	National Meteorological Agency, Ethiopia
LU/LC	Land sat satellite images (1985, 1999, 2009 and 2019)	USGS (https://earthexplorer.usgs.gov/)
C factor	Land use land cover classes	Own analysis
P factor	LU/LC class maps and slope map	Own analysis

3.3.1. The RUSLE-factors

The processes of soil erosion are influenced by two major factors, geomorphology (physical features) and hydrology of the catchment (Aga et al., 2019; Marttila and Kløve, 2010). The spatial variability of soil erosion is due to the variation in climatic conditions, physiographic characteristics of an area and anthropogenic activities (Shen et al., 2017). Depending on their data requirements and applicability, different methods (models) were developed to estimate soil erosion risk in different parts of the world. In this study RUSLE with GIS and remote sensing was employed to quantify annual soil loss (Auerswald et al., 1992). This model is one of the empirical models and widely applied in regions where data sources are scarce (Hurni, 2002; Bewket and Teferi, 2009). The effectiveness of this model has been confirmed by previous studies (Markose and Jayappa, 2016; Rozos et al., 2013). Soil loss estimation using RUSLE in the GIS environment involves the integration of five different parameters (R, K, LS, C and P) as shown in equation (12) below (Renard, 1997; Sheikh et al., 2011). Hence, the accuracy of soil erosion modelling using RUSLE is influenced by proper estimation of soil erosion controlling factors and their data sources. However, this method only estimates soil loss from sheet and rill erosion, not considering gully and stream bank erosion. In addition, it is incapable of estimating SDR and SY. Because of this, other methods were applied with this model to estimate SDR and SY.

$$A = R*K*LS*C*P \qquad \text{Eq (12)}$$

Where, A is the annual soil loss (metric tons/ ha/year); R is the rainfall erosivity factor (MJ mm/ h/ ha/ year); K is the soil erodibility factor (metric tons/ ha/MJ /mm); LS = slope length and steepness factor (dimensionless); C is land cover factor (dimensionless); and P is conservation practice factor (dimensionless).

Rational of using RUSLE for soil erosion modelling

The Revised Universal Soil Loss Equation (RUSLE) was developed by Weltz *et al.* (1998) to address the limitations of USLE by accounting for the soil erosion factors. RUSLE modifies R_factor to include the impacts of raindrops on the detachment of sol particles in flat slopes. Improvement in slope length and steepness (LS) factor in RUSLE is a major change that accommodates complex slopes. Moreover, modifications were considered in the C_factor (contribution of residue considered) and P_factor (strip cropping included). RUSLE can also incorporate converging and diverging terrain and areas of net sediment accumulation. All these improvements in soil erosion factors enable users to apply this model in the study of soil erosion risk and sediment delivery in different catchments having various spatial scales and complex topography.

Rain fall erosivity (R) factor

Rain fall erosivity is one of the factors of soil erosion and it is an index that quantifies detached and transported sediment from sheet and rill erosion by rainfall/runoff (Wischmeier and Smith, 1978; Woldemariam and Harka, 2020).This parameter can be estimated by considering the kinetic energy of the storm and the maximum 30-minute intensity (Wischmeier and Smith, 1978). However, meteorological stations lack long-term data of 30-minute intensity and because of this gap other empirical equations that correlate mean annual rainfall and R factor were developed (Manaouch et al., 2021). For this study, rainfall data of the watershed were obtained from National Meteorological Service Agency. Data from four stations (Bichena, Kuy, Robegebeya and Yetmen) were used to calculate R- factor values of the study area (Fig 3.4 and 3.5; Table 3.7). Monthly rainfall records from these stations covering the period of 45 years (1974- 2019) were used. The mean annual rainfall was first interpolated to generate continuous rainfall data for each grid cell by “3DAnalyst Tools Raster IDW Interpolation” in Arc GIS environment. Then, the erosivity (R) factor value was calculated using equation (13) suggested by Hurni (1985) for the Ethiopian conditions.

$$R = -8.12 + 0.562P \qquad \text{Eq (13)}$$

Where R is the rainfall erosivity factor and P is the mean annual rainfall (mm).

Table 3.7. Average annual rainfall and R- factor values for stations in and near the Suha watershed

Rainfall Station	Geographical location		Elevation(m)	Mean annual rainfall (mm) and R- factor values							
	Latitude	Longitude		1974-1985		1985- 1999		1999 - 2009		2009 - 2019	
				RF (mm)	R-factor	RF (mm)	R-factor	RF (mm)	R-factor	RF (mm)	R-factor
Bichena	38.204	10.445	2532	-	-	-	-	-	-	1322	735
Debre Markos	37.739	10.326	2446	1336	743	1327	737.6	1416	788	1341	746
Dejen/ Yetnora	38.151	10.171	2448	1036	574	1255	697	1410	784	1213	673
Robege beya	37.87	10.55	2940	-	-	1391	774	1505	838	1396	777
Kuy	37.594	10.3	2420	-	-	-	-	-	-	1316	731

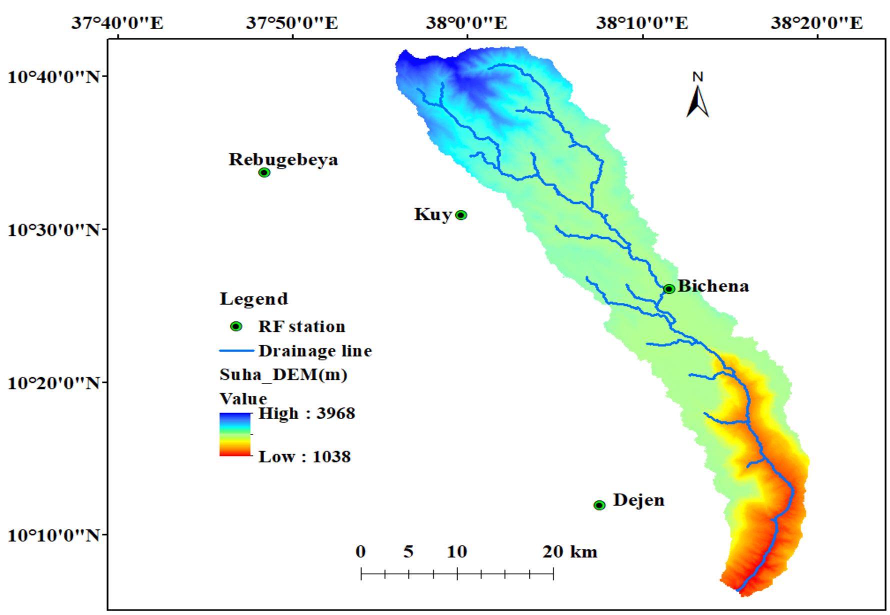


Fig 3.4. Rain gauge stations found in and around the Suha watershed

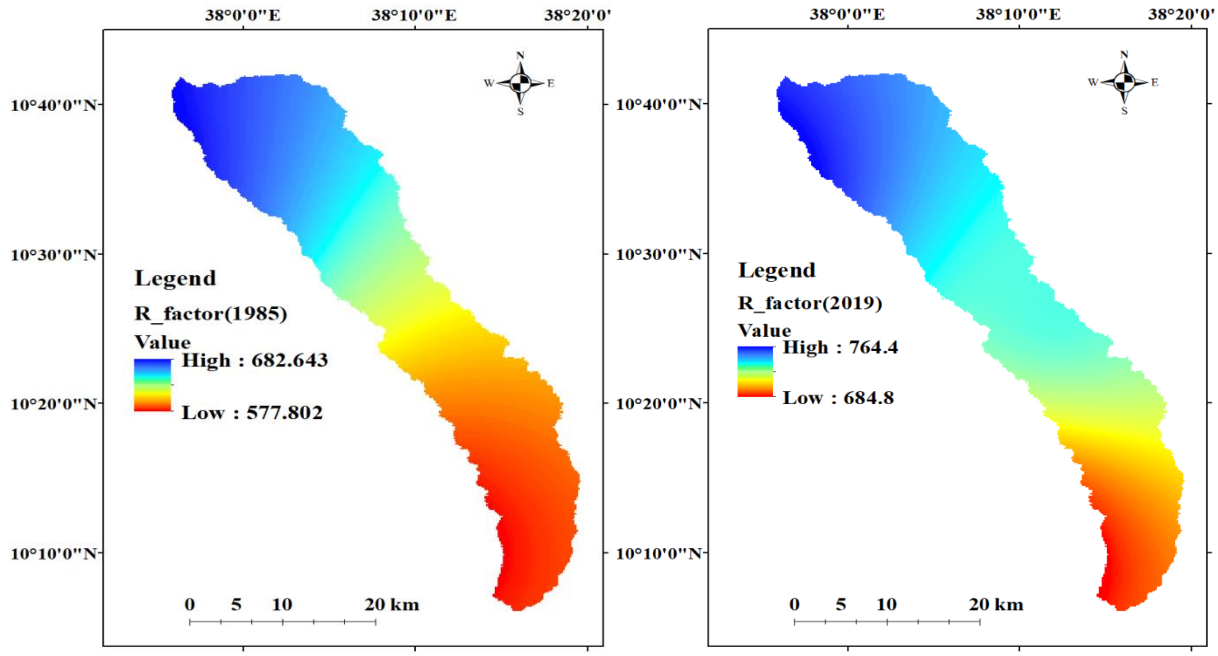


Fig 3.5. R_factor maps of a) 1985 and b) 2019

Soil erodibility (K) factor

Soil erodibility is the most important factor that determines soil erosion and it signifies the inherent characteristics of the soil to erosion (Manaouch et al., 2021). The severity of soil erosion and its spatial variability is due to the variation in soil texture, OM content, aggregate stability and permeability (Uddin et al., 2019; Panagos et al., 2014). It is an important parameter in understanding soil erosion mechanisms (processes). Different methods can be applied in estimating soil erodibility; which includes: examining soil properties and using different experiments like score, rainfall simulation and plot experiments (Song et al., 2005). An equation was developed to measure the K- factor using soil texture and organic matter as data input (Pham et al., 2018).

$$\mathbf{K = 2.1 * 10^{-6} * M^{1.14} * (12 - OM) + 0.325 * (P-2) + 0.025 * (S - 3)} \quad \text{Eq (14)}$$

Where, M = (percentage silt + percentage very fine sand) (100 percent clay); OM = the percentage of organic matter content; P = profile permeability; and S = structural class.

However, in data-scarce regions, like Ethiopia, data from literature can be applied. In this study, the calculation of soil erodibility involved the identification of soil types and estimating corresponding K- factor values. First, soil types were extracted from Abay Basin master plan

(MoWR, 2011) using "Spatial Analyst Tool'' in GIS environment and six soil types were identified. K-factor values, which are recommended for Ethiopian conditions, were adopted from Hurni (1985) and Hellden (1987) for each soil type. Using these values raster maps were prepared and reclassified for further analysis (Fig 3.6). The soil types and their erodibility factors are presented in (Table 3.8).

Table 3.8. Soil types and adopted K - factor values in Suha watershed

Soil types	Area (ha)	proportion in %	K - values	References for (K - values)
Eutric Cambisols	7680.3	9.6%	0.2	
Eutric Leptosols	10829.1	13.5%	0.2	Hurni (1985); Hellden
Eutric Vertisols	46123.4	57.4%	0.15	(1987); SCRP (1996)
Haplic Nitosols	210.5	0.3%	0.25	
Haplic Luvisols	4900.6	6.1%	0.2	
Rendzic Leptosols	10599	13.2%	0.2	
Total	80343	100.0		

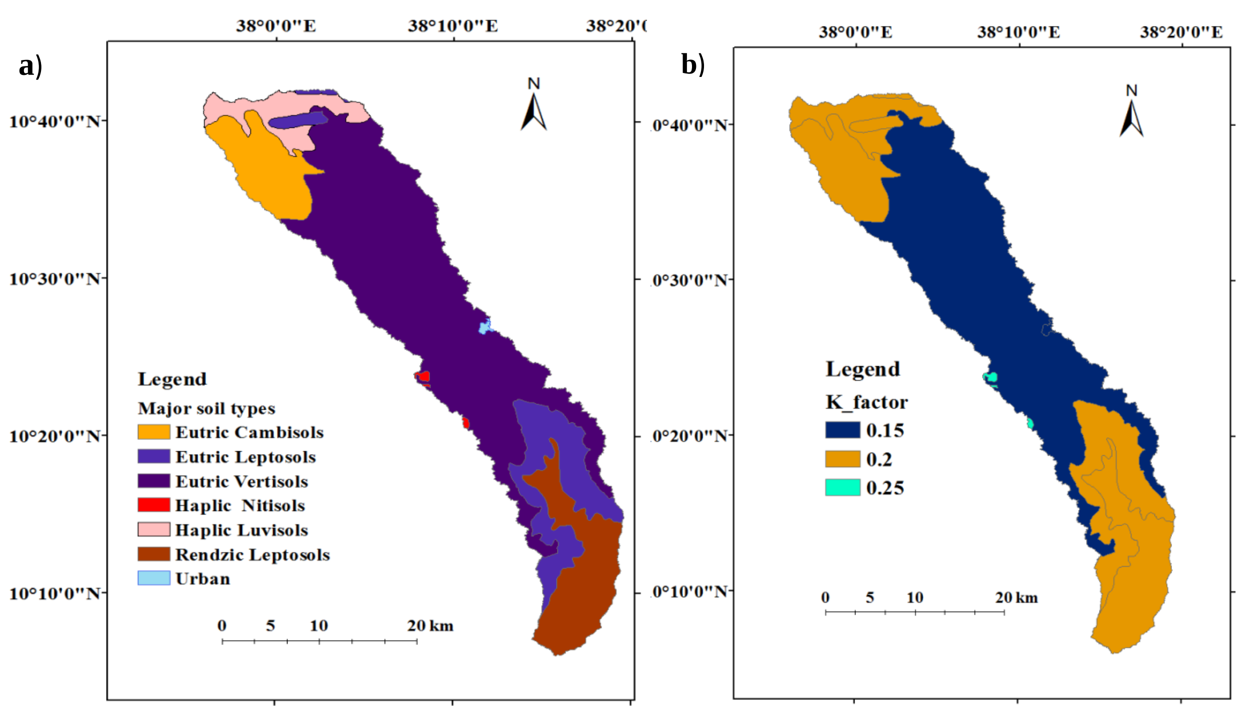


Fig 3.6. Map of a) soil type and b) K_factor for Suha watershed

Topographic (LS) factor

One of the most crucial variables affecting soil erosion rate is topography, which quantifies the combined impact of slope length (L) and steepness (S) on soil erosion. Slope length (L) and slope steepness (S) characterize the topographic feature of an area. LS factor is the ratio of soil loss from a given area to that of RUSLE standard plot with length (22.13 m) and steepness (9%), while maintaining other factors constant (Renard et al., 2011). The values of the LS factor are influenced by the landforms of the catchment, as well as the flow direction and accumulation. Various algorithms have been developed to estimate LS factor values. The first algorithm was developed by Wischmeier and Smith (1978) (Equation 15).

$$\mathbf{LS}=(\boldsymbol{\lambda}/\mathbf{22.13})^{\mathbf{m}} * (\mathbf{65.4} * \mathbf{sin^2\beta} + \mathbf{4.5} * \mathbf{sin\beta} + \mathbf{0.00654}) \qquad \textbf{(Eq15)}$$

Where, B is the slope angle in radiance and m is the power exponent. The value of m is between 0.2 and 0.5 depending on the slope of the catchment.

Another method developed by McCool et al. (1989) also considered slope length and slope of the catchment to calculate LS_factor values (Equation 16).

$$\mathbf{LS} = (\boldsymbol{\lambda}/\mathbf{22.13})^{m} * \begin{cases} (\mathbf{10.8sin\beta + 0.03})\ \mathbf{if}\ \boldsymbol{\beta} < 0.09 \\ (\mathbf{16.8sin\beta - 0.5})\ \mathbf{if}\ \boldsymbol{\beta} > 0.09 \\ (\mathbf{3sin^{0.8}\beta + 0.56})\ \mathbf{if}\ \boldsymbol{\lambda} < 4.5m \end{cases} \qquad \text{Eq(16).}$$

Moore and Burch (1986a) also modified previously developed methods.

$$\mathbf{LS} = (\mathbf{AS/22.13})^{\mathbf{m}} * (\mathbf{sin\beta/0.0896})^{\mathbf{n}} \qquad \text{Eq (17)}$$

Where, *m* = 0.4 (values ranged from 0.4 to 0.6) and n= 1.3 (values ranged from 1.22 to 1.3)

Even though different algorithms were developed so far, most of them are site-specific and LS_ values are highly variable across regions. In this study, the method proposed by Hurni (1985) for the Ethiopian condition was adopted to generate LS_factor values (Table 3.9). A digital elevation model (DEM) of 30 m x 30m spatial resolution was used to generate the slope of the watershed and slope length using “Spatial Analyst Tool Surface Slope” in Arc GIS 10.4 environment.

.

The approach was, first to analyze the slope length and slope of the watershed then reclassified based on factor_L and factor_S values and finally superimpose these two maps to get LS_ factor value which gave us reasonable value compared to other algorithms (Fig 3.7).

Table 3.9. Factor_L and Factor_S values derived from the slope length and slope of the Suha watershed

Slope length (m)	Factor _L	Slope (%)	Factor_S	Reference
< 5	0.5	< 5	0.4	
5-10	0.7	5-10	1.0	Hurni, 1985
10-20	1.0	10-15	1.6	
20-40	1.4	15-20	2.2	
40-80	1.9	20-30	3.0	
80-160	2.7	30-40	3.8	
160-240	3.2	40-50	4.3	
> 240	3.8	> 50	4.8	

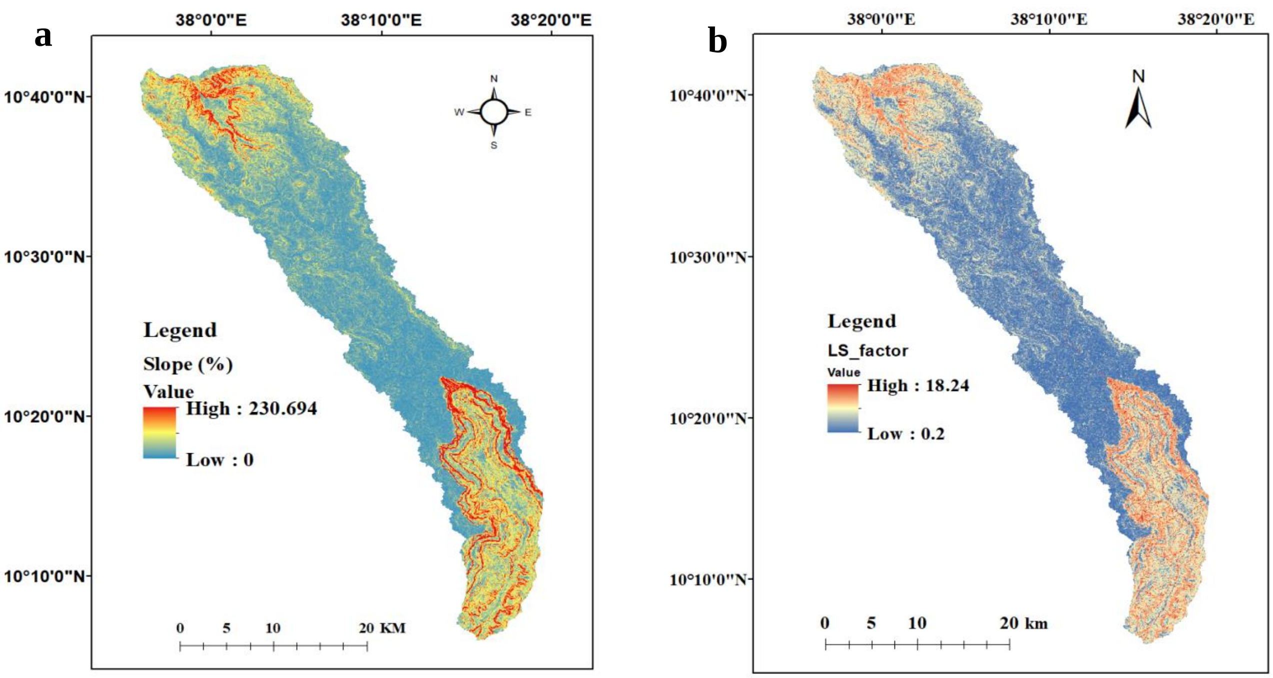


Fig 3.7. Slope map (a) and LS_factor map (b) of Suha watershed

Land cover factor(C)

This factor represents the extent of land cover by different vegetation types, from dense forest to annual crop cover. The values of this parameter ranged from 0 to 1, in which the lower value represents dense forest cover and the higher value represents bare lands (Erencin et al., 2000).Values can be estimated using various techniques; one of the methods used is the application of the Normalized Difference Vegetation Index (NDVI) (Yan et al., 2020; Karaburun, 2010). To estimate C factor values, the primary task was computing NDVI values from Landsat satellite images using Near Infrared (NIR) and Red (R) bands which have different spectral reflectance (Arekhi et al., 2012). The minimum and maximum values of NDVI were -1 and 1 respectively. The lower value represents bare land and the largest value for land covered with dense forest. NDVI and C factor values were computed as follows.

$$\mathbf{NDVI} = \frac{\mathbf{NIR - RED}}{\mathbf{NIR + RED}} \qquad \text{Eq (19)}$$

$$\mathbf{C} = \left(\frac{\mathbf{1 - NDVI}}{\mathbf{2}}\right) \qquad \text{Eq (20)}$$

Another approach (adopting C-factor values from literature), commonly used by other researchers in Ethiopian conditions, was applied for this study. For each land use/ cover class, C_factor values were assigned by adopting the recommendation of Hurni (1985); Degife et al. (2021); Gashaw et al. (2017) for the Ethiopian condition (Fig 3.8); other scholars also used a similar approach (Degife A. et al., 2021; Fenta et al., 2021). Then, raster maps were prepared to integrate with other factors in RUSLE. Six types of land use and land cover classes were identified in the study area; their spatial and temporal variability and magnitude of changes are described in (Table 3.10).

Table 3.10. Land use/cover classes and their corresponding C- factor values in the Suha watershed

LULC class	Area (ha)				adopted C-factor values	
	1985	1999	2009	2019	values	Reference
Agricultural land	44313.9	51938.9	57523.8	59731.5	0.25	Hurni, 1985; Degife et al., 2021; Gashaw et al., 2017
Grazing land	25762.2	18272.3	11329.5	7193.8	0.014	
Forest land	668.0	543.6	883.2	1076.5	0.001	
Shrub land	7441.3	6592.5	4364.5	3897.1	0.02	
Bare land	1417.7	1967.4	4854.5	6714.9	0.6	
Built up area	725.7	1016.9	1376.4	1719.7	0.05	

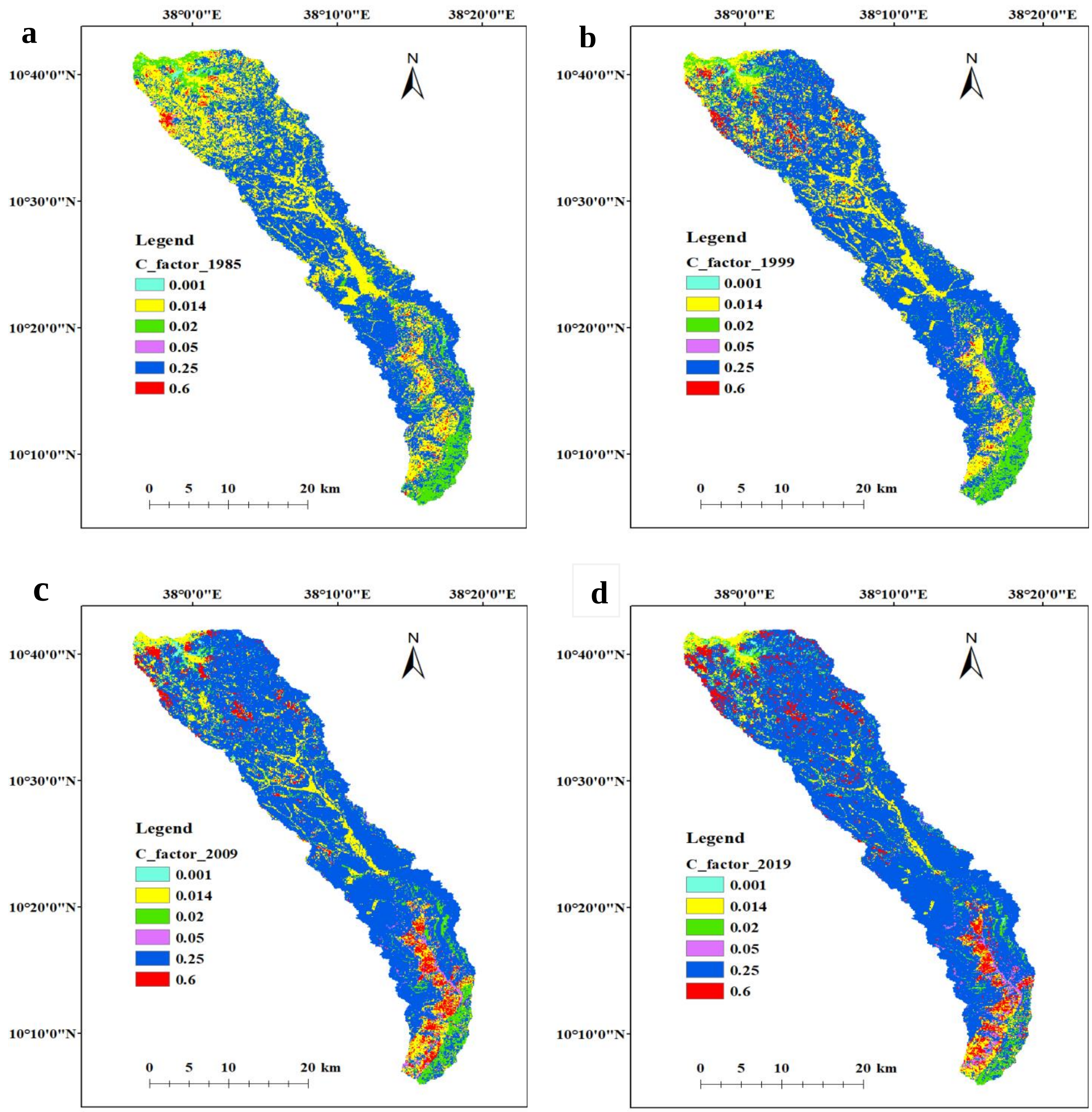


Fig 3.8. C-factor maps of LULC classes a) 1985, b) 1999, c) 2009 and d) 2019

Conservation or management factor (P - factor)

The application of conservation strategies can effectively mitigate the primary driving forces of soil erosion, the velocity and concentration of runoff (Renard, 1997). The p-factor serves as an indicator of the impact of soil and water conservation technologies on soil erosion. This parameter represents the ratio of soil loss from managed fields to bare land or cultivated fields that are sloping up or down (Morgan, 1996; Wischmeier and Smith, 1978). The values of the p-factor range from 0 to 1, with the lowest value indicating well-managed fields and the highest value representing unmanaged fields (Morgan, 1996). To calculate the RUSLE - P factor values, two data inputs are required: land use type or management and the slope of the catchment (Behera et al., 2020). In this particular study, the P_factor values recommended for Ethiopia conditions (Hurni, 1985) were adopted and assigned to each land use/cover class to generate P_factor raster maps for further analysis (Fig 3.9 and Table 3.11). Other researchers (Atoma et al., 2020; Tadesse et al., 2017; Tamene et al., 2017) have also utilized the same approach to assess soil erosion risk in Ethiopia.

Table 3.11. P_factor values adopted for different land use classes in the Suha watershed

Land use land cover class	P_factor value	Reference
Agricultural land	0.9	Moisa et al. (2021); Atoma et al. (2020)
Grazing land	0.6	Moisa et al. (2021) ; Atoma et al. (2020); Tadesse et al. (2017)
Forest land	0.53	Moisa et al. (2021); Atoma et al. (2020)
Shrub land	0.63	Moisa et al. (2021)
Bare land	1	Modified from Moisa et al. (2021)
Built up area	0.7	Modified from Moisa et al. (2021)

3.3.2. Sediment delivery ration (SDR) and Sediment yield (SY) estimation

Since the RUSLE model is incapable of estimating SDR and SY, other methods are employed and integrated with RUSLE. So far various methods/ models have been developed and used to quantify SDR in different parts of the world (Bekele and Gemi, 2021). Differences in biophysical, climatic, and hydrologic factors and the availability of the data force researchers to develop different approaches to estimate SDR for a given catchment. The results of these studies are highly variable across regions/ catchments. For data-scarce regions other approaches were developed and used by different researchers. Since measured sediment data is scarce in the study area, the following two methods were employed and the results of these methods were compared.

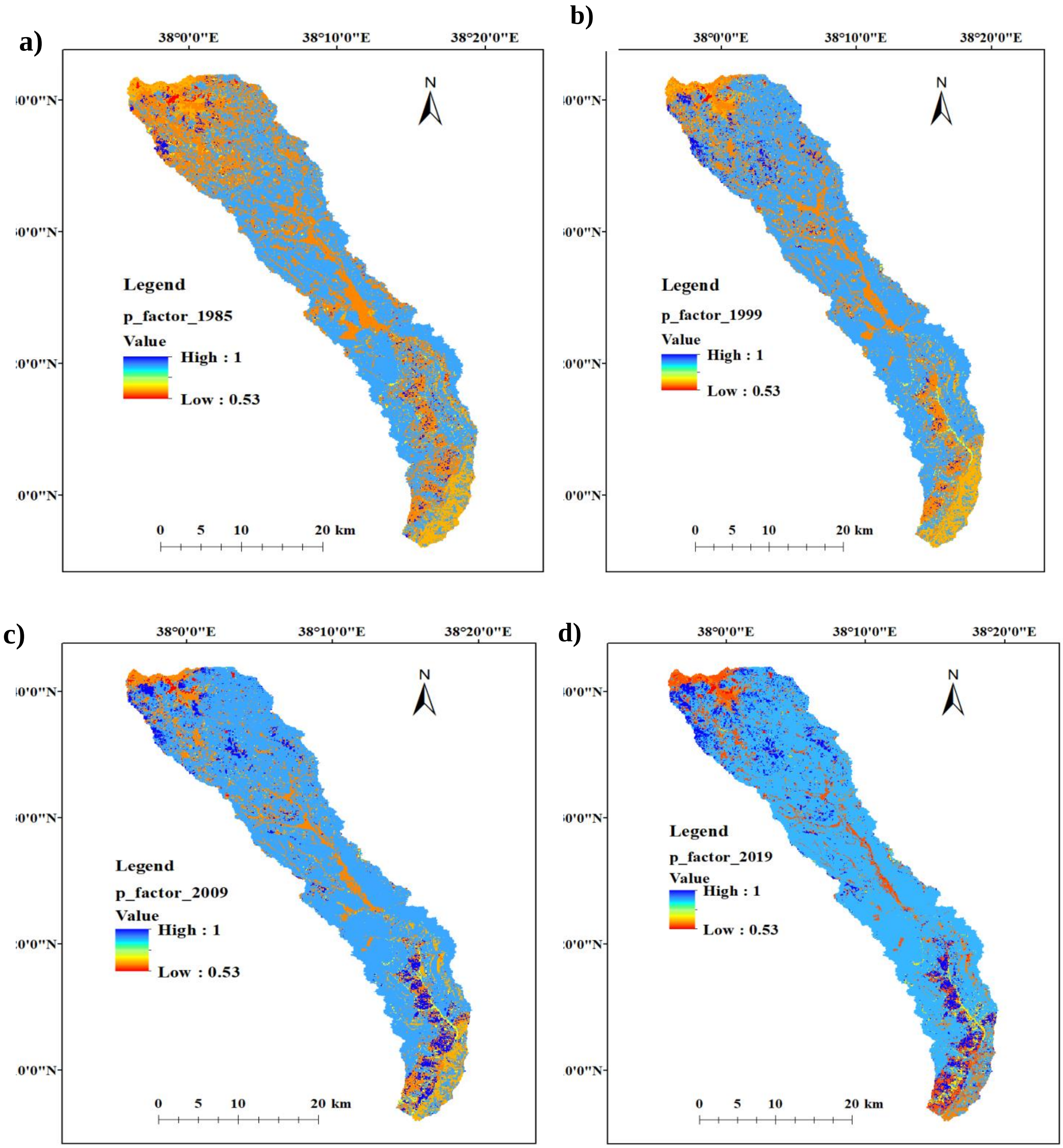


Fig 3.9. P_ factor maps of a) 1985; b) 1999; c) 2009 and d) 2019 for Suha watershed

Sediment delivery ratio (SDR) is the proportion of eroded sediment that reaches the outlet of the catchment to the total amount of eroded sediment (Julien and Frenette, 1998). It is an input data used to estimate sediment yield (SY). The amount of sediment reaching the outlet of the catchment is governed by the physiographic characteristics of the catchment (catchment area, soil type, topography, runoff and stream channel slope (Aga et al., 2019). In this study, SDR and SY were estimated using drainage area as data input (equation 21). The same approach has been employed by different scholars (Bekele and Gemi, 2021; Sharda and Ojasvi, 2016).

$$\mathbf{SDR = A^{-0.2}} \qquad \text{Eq (21)}$$

Where, SDR = sediment delivery ratio and

A = area of the catchment in square kilometers (km^2).

Sediment delivery ratio (SDR) estimation using stream channel bed slope

Another method that can be used to estimate SDR in data-scarce regions is using stream channel bed slope as data input in Arc GIS environment. This approach has been applied in different catchments of Ethiopia where data availability is challenging (Zerihun et al., 2018). In this study, DEM with 30m*30m resolution was used as data input for this purpose. Flow direction, accumulation and stream network were analyzed from this DEM. In addition, the HEcGeoHMS tool in Arc GIS was applied to calculate the stream channel bed slope using DEM and flow path.

$$\mathbf{SDR} = \mathbf{0.627} * (\boldsymbol{SCS})^{\mathbf{0.403}} \qquad \text{Eq (22)}$$

Where, SDR is the sediment delivery ratio and SCS is the stream channel bed slope (%).

Sediment yield (SY)

It is the amount of sediment that leaves the watershed (Sharp et al., 2018).The scarcity of measured sediment data is the main problem in the study of sediment dynamics. In this case, another approach, integrating gross soil loss raster layers with SDR of the watershed using spatial analyst tool in ArcGIS and computing SY is applicable (Equation 23) (Mutua et al., 2006). The same approach was applied in this study.

$$\mathbf{SY = E * SDR} \qquad \text{Eq (23)}$$

Where, SY = Sediment yield (ton/yr);

SDR = the proportion of sediment reaching in the outlet of the watershed

E = Total soil loss (ton/yr) of the watershed

Sediment retention

It refers to the amount of sediment that is deposited in the landscape position along the flow path of the sediment (sediment deposited downstream of the catchment/not reaching the outlet of the catchment).This data is required for identifying sediment sources and designing management strategies. The amount of sediment retained in pixel i, Ei of the catchment can be calculated using equation (24):

$$\mathbf{E'_i = RUSLE'i} * (\mathbf{1 - SDRi}) \qquad \text{Eq(24)}$$

3.3.3. Identification of soil erosion hotspots

To delineate sub-watersheds, Arc Hydro Tools in ArcGIS10.5 environment was applied. Step by-step processes were followed using a 30*30 meter resolution of SRTM DEM as an input in the terrain processing of Arc Hydro tools. Fill sink, flow direction, flow accumulation, stream definition, stream segmentation, catchment gird delineation, catchment polygon processing, drainage line processes and drainage point processing were the steps followed. Based on the analysis result, the study catchment was classified in to 24 sub-watersheds (Fig 3.10).

Prioritization of the sub-watersheds was carried out based on the magnitude of the rate of soil erosion, which was obtained by superimposing a map of annual soil loss of the watershed with sub-watershed maps in Arc GIS. The application of zonal analysis based on the hydrological response unit (HRU) by grouping identical hydrological responsive areas, is also becoming a useful approach to delineate landscapes having similar soil erosion rates (Kumar and Mishra, 2015). This task is essential for resource use efficiency and effectiveness of watershed development. Soil and water conservation activities demand huge investments (resource and labor-intensive), implying that limiting available resources for priority areas is essential. The general framework of this study is highlighted in (Fig 3.11).

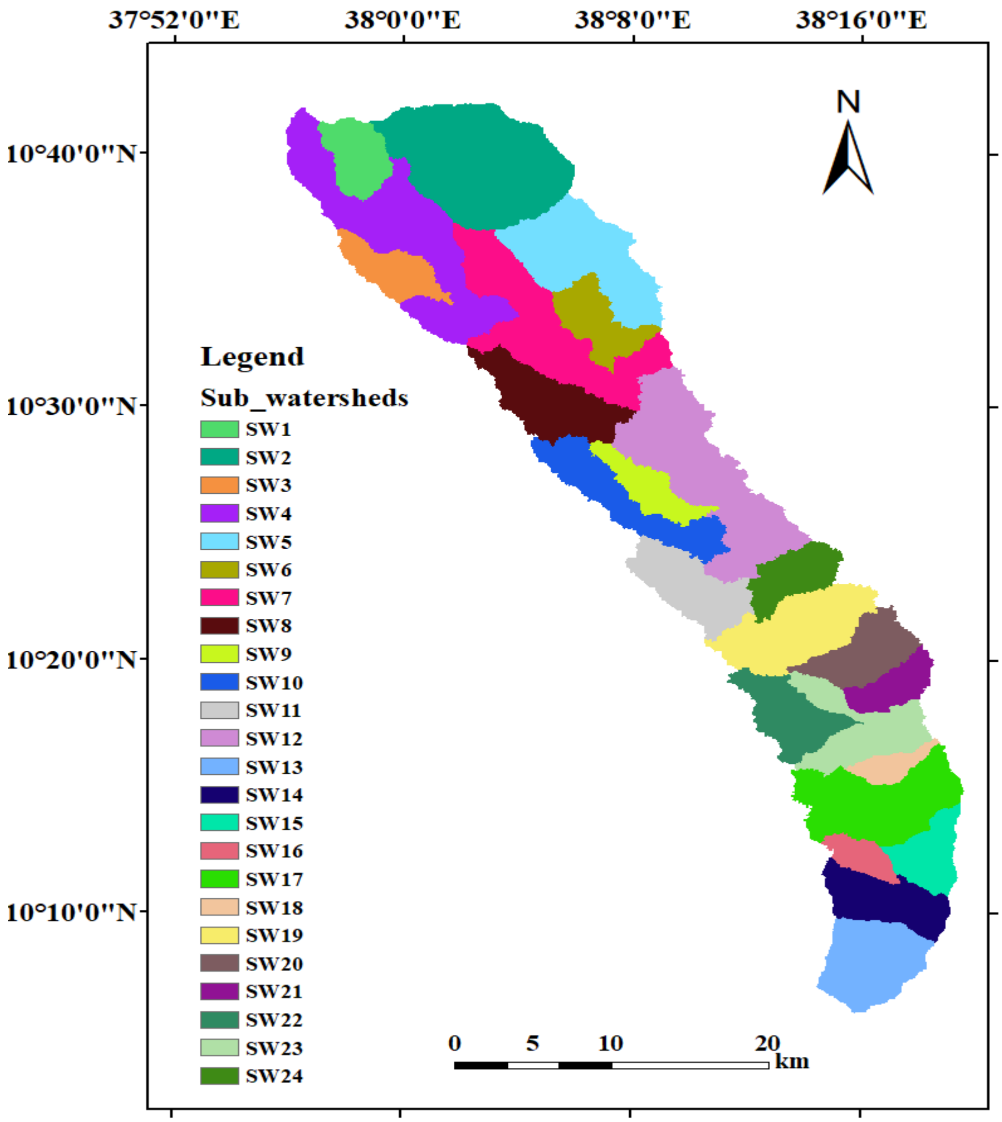


Fig 3.10. Sub watersheds of the study area

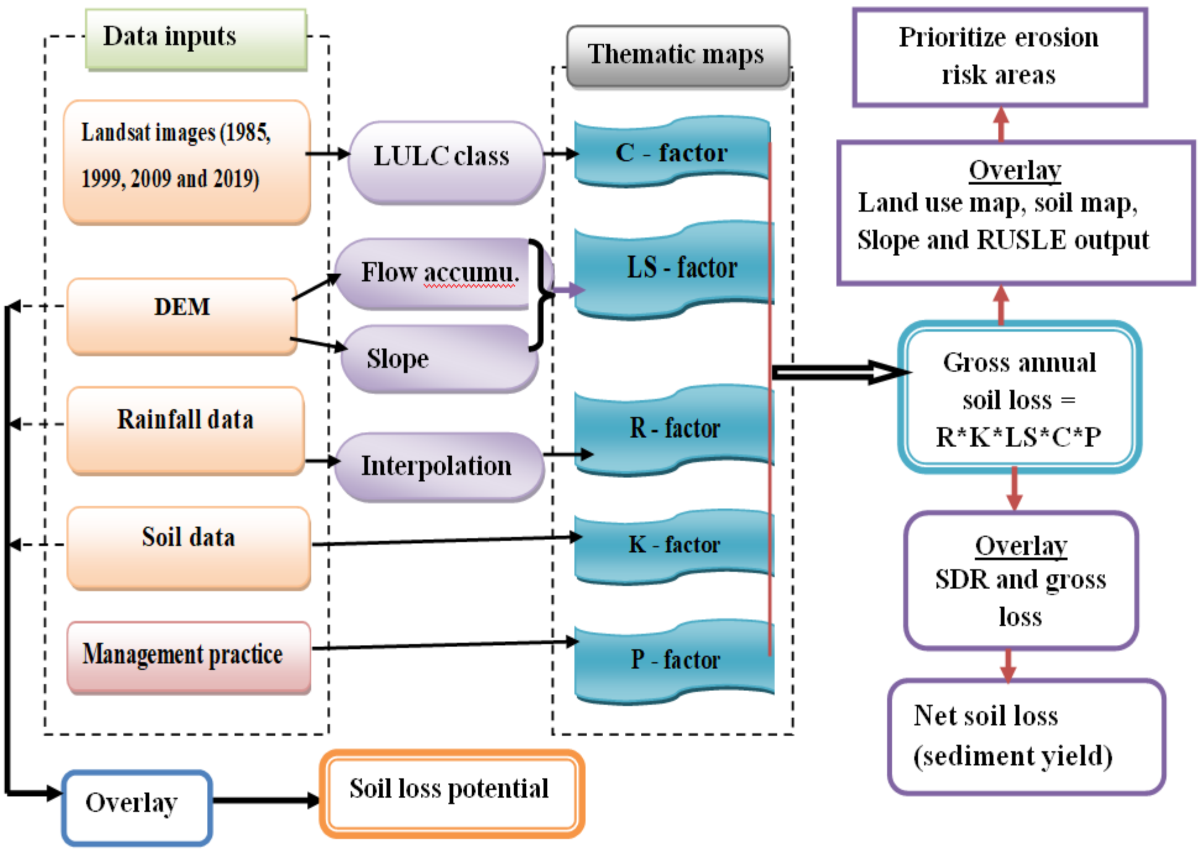


Figure 3.11. Methodological framework used to estimate soil loss and sediment yield

3.4 .Monitoring soil quality

3.4.1. Soil sampling

Prior to soil sample collection, the watershed has been stratified in to three strata based on its altitudinal gradient using the digital elevation model (DEM) of the watershed. These strata were classified as upper (2700-3900 m.a.s.l), middle (2400-2700 m.a.s.l) and lower (1080-2400 m.a.s.l) parts of the watershed. Then, tentative soil sampling sites were fixed on the land use/ land cover maps along the toposequence of the watershed. In addition, a reconnaissance field survey and discussion with development agents were carried out to get a general over view of the watershed. A totally of 27 composite surface soil samples (0-30 cm) were collected using a soil auger in three replications from adjacently located land use systems (agricultural land, grazing land and forest land) having similar slope gradient soil type and climatic condition from each stratum (upper, middle and lower part of the watershed). Ten to fifteen primary samples were collected to prepare one composite sample. Frame work of the methodology was highlighted in (Fig 3.12). Soil quality indicators including soil particle size distribution, Index of soil aggregate stability (ISS), soil water contents, soil pH-H_2O, organic carbon (OC), total nitrogen (TN), carbon to nitrogen ration (C:N), available phosphorus (Av.P), Exchangeable bases (Na^+, K^+, Ca^{2+} and Mg^{2+}), PBS and cation exchange capacity (CEC) were considered in this study. Geographical locations and elevation of sampling sites were gathered using a global positioning system (GPS).

3.4.2. Laboratory analysis

One kg of composite samples were prepared, packed in plastic bags, labeled and taken to Ethiopia Construction, Design and Supervision Corporation for analysis. Soil samples were air-dried, crushed and sieved through a 2 mm and 0.5 mm mesh sieve to remove coarser particles. Standard laboratory procedures were followed in the analysis of soil physical and chemical quality indicators.

Particle size distribution was determined by the Boycouos hydrometric method (Bouyoucos, 1962) after destroying organic matter (OM) using hydrogen peroxide (H_2O_2) and dispersing the soils with sodium hexameta phosphate (NaPO3). The index of soil aggregate stability (ISS) was determined using equation (2) as described by (Pieri, 1992). Determination of soil water contents, FC and PWP were determined at 1/3 bar and 15 bar respectively by pressure membrane suction method (Estefan et al., 2013).Then, plant available water holding capacity (AWHC) was calculated using equation (25)

$$\mathbf{ISS} = \frac{\mathbf{OC} * \mathbf{1.724}}{(\mathbf{L}+\mathbf{A})} * \mathbf{100} \qquad \textbf{Eq. (25)}$$

Where, ISS= index of soil aggregate stability; OC= organic carbon; L= silt proportion and A= clay proportion.

$$\mathbf{AWHC}\ (\%) = \mathbf{FC} - \mathbf{PWP} \qquad \textbf{Eq. (26)}$$

Where, AWHC, available water holding capacity; FC, field capacity and PWP, permanent wilting point

Soil pH was determined in water (pH- H2O) using a 1:2.5 soil to water solution ratio with a pH meter as outlined in Van Reeuwijk (2006). The Walkley and Black wet digestion method was used to analyze soil OC content (Van Ranst, 1993). Total N was analyzed using the Kjeldahl digestion, distillation and titration method as described by Bremner and Mulvaney (1982). Available soil P was analyzed according to the standard procedure of Olsen et al. (1954).

Exchangeable bases (Ca^{2+}, Mg^{2+}, K^{+} and Na^{+}) were determined by ammonium acetate (1N NH4OAc) at pH 7.0. Exchangeable Ca^{2+} and Mg^{2+} in the extracts were measured using atomic absorption spectrophotometer, while K^{+} and Na^{+} were determined by a flame photometer (Van Ranst 1993). Cation exchange capacity (CEC) was estimated titrimetrically by distillation of ammonium that was displaced by sodium from NaCl solution (Chapman, 1965).Percent base saturation (PBS) was calculated by dividing the sum of basic cations (Ca2+, Mg2+, K+, and Na+) by the CEC of the soil and multiplying by 100.

$$\mathbf{BS\% = \frac{Ca^{2+} + Mg^{2+} + K^{+} + Na^{+}\ (cmol(+)/kg)}{CEC\ cmol(+)/kg} \times 100} \qquad \text{Eq. (27)}$$

Where, BS% is base saturation in percent (PBS)

$$\mathbf{ESP = \frac{Na^{+}\ (cmol(+)/kg)}{CEC\ cmol(+)/kg} \times 100} \qquad \text{Eq. (28)}$$

Where, ESP is exchangeable sodium percentage

3.4.3. Soil quality deterioration index (SQDI)

Soil quality deterioration index (SQDI) for each soil quality indicator under different land use systems and elevation gradient was computed using an undisturbed ecosystem (forest land) as a reference and evaluation of soil parameters of other land systems againist this reference as described by (Abera and Assen, 2019; Gui et al., 2009). Deterioration index (DI) Values were computed by taking the difference between the mean values of quality indicators in the given land use and the mean values in the referenced land use system. From the total of 19 soil quality indicators, 13 indicators (clay, silt, ISS, AWC, pH, OC, TN, Av.P, Ca+, Mg2+, Na+, K+ and CEC) were selected using an expert opinion (Anderson, 2002) to compute SQDI values. Most of these indicators are sensitive to land use change/ land management (Abera and Assen, 2019).

Individual soil quality indicators couldn’t give a proper estimation of soil quality and hence the combination of soil quality indicators is required. So far different approaches were developed to assess soil quality. However, there is no universally accepted technique that could be applied to evaluate soil conditions in various ecosystems (Mukherjee and Lal, 2014; De Paul and Lal, 2016). Recently, Soil Management Assessment Framework (SMAF) approach is becoming widely accepted and applied in different countries/regions (Andrews et al., 2004) (equation 29).

$$\mathbf{SQI} = \sum_{i=1}^{N} \mathrm{WiXi} \qquad \text{Eq(29)}$$

Where, SQI is the soil quality index; W is the normalized indicator; X is the indicator score; i is a soil property and n, is the number of soil properties.

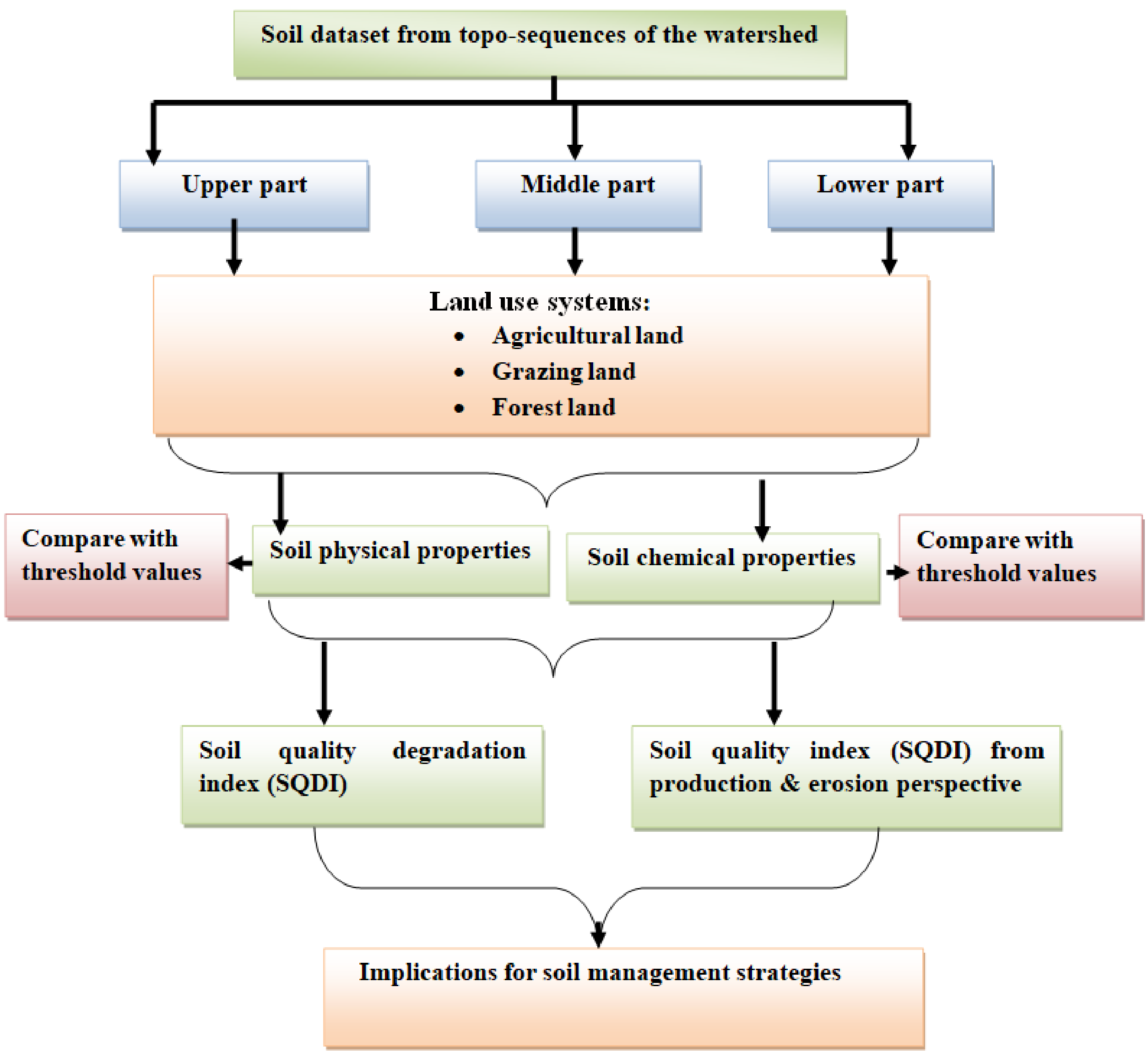


Fig 3.12. Framework of the methodology for soil quality assessment

3.5. Quantification of soil nutrient flows and balances in cereal-based agroecosystems

3.5.1. Selection of case study farms

Two different agroecologies (highland and lowland) were selected to study soil nutrient flows and balances. The representativeness of soil type, and wealth class of farmers were considered for the selection of the case study farms. For a detailed analysis of soil nutrient flows and balances, a total of 9 farms (3 from the highland and 6 from the midland) were selected by considering the wealth status (rich, medium and poor) of the households. The analysis of N and P balances was conducted based on nutrient fluxes (input and output flows) which enable us to identify areas of nutrient depletion and management options. Data of farm management strategies, type and amounts of inputs used (mineral and organic inputs), crop harvested products (yield and residue) and crop residue management strategies were collected through field surveys and interviews with farmers. Data on crop harvests were substantiated with field measurements. The methodological approach used in this study was presented in (Fig 3.13).

3.5.2. Quantification of soil nutrient inflows and out flows

To quantify the nutrient flows and balances of macronutrients (N, and P), four input flows and five output flows were considered (Tables 3.12). Estimation and quantifications of soil nutrient flows and balances were based on the data obtained from field measurements, field surveys (interviews) and data from the literature. The values of soil nutrient balances are indicators of either nutrients are accumulated in the soil (positive balances) or the occurrence of nutrient depletion (negative balance) (Elias et al., 2004; Haileslassie et al., 2005; Nandwa and Bekunda, 1998).

Table 3.12. Nutrient inflow processes, data sources and methods applied to quantify nutrient Contents (adapted from Elias et al., 1998; Lesschen et al., 2007; Lewoyehu et al., 2020)

Inflows	Nutrients	Data types/sources	Method of quantification
Mineral fertilizer (IN1)	N and P	Type of fertilizes applied ; nutrient contents of fertilizers	Field survey and secondary data
Organic input (IN2)	N, and P	Amount of organic inputs applied Nutrient contents of organic input	Field survey and secondary data
Atmospheric deposition (IN3)	N, and P	Annual rainfall	Regression equations (transfer functions)
N- fixation (non-symbiotic) (IN4b)	N	Rainfall; amount of N- fixed	Regression equation (transfer functions)

Table 3.12. Continued. Nutrient outflow processes and methods

Outflows	Nutrients	Data types/sources	Method of quantification
Crop harvest (OUT1)	N, and P	Crop yield; nutrient content of the grain	Field measurement and secondary data
Crop residues (OUT2)	N, and P	Removed crop residues from the field; nutrient contents of residue	Field measurement and secondary data
Leaching (OUT3)	N	Annual rain fall; chemical and organic inputs; soil fertility class	Regression equations (Stoorvogel and Smaling, 1990)
Gaseous loss (OUT4)	N	Annual rain fall; nutrient content of chemical and organic inputs; SOC	Regression equation (FAO, 2003)
Erosion (OUT5)	N, and P	DEM, soil erodibility, rainfall erosivity, slope length and steepness, LU/LC and soil management practices	RUSLE model

3.5.2.1. Quantification of nutrient input flows

Chemical fertilizers are the main sources of nutrient inputs for N and P nutrients. The data on the type and amounts of mineral fertilizers applied in each farm were collected through field surveys, interviews with farmers and secondary data (district data of mineral fertilizers used). Based on these data sources, mineral fertilizers commonly used were NPSB and Urea. The chemical composition of NPSB is 18.9% N, 37.7% P_2O_5, 6.95% S and 0.1% B, whereas, Urea contains only 46% of N. The amount of fertilizers applied varies across the field due to the difference in soil type, farming systems and economic capability of the household to purchase these inputs. The amount of N and P supplied to each farm was calculated by multiplying the amount of applied chemical fertilizers and the corresponding compositions of N and P (Elias et al., 1998; Mulualem et al., 2021).

The amount of nutrients supplied in to the farming system from the organic inputs can be estimated by multiplying the dry matter of this material by the nutrient contents (Aticho et al., 2011). The data of organic inputs (N2) were collected through field surveys and interviews with the farmers. In the study area, there was no experience of applying organic inputs to the farmlands which were found far from the residential area (there has been some experience of using it around homesteads). Because of this, nutrient input from this component was not considered.

The amount of wet deposition of nutrients depends on the amount of annual rainfall. These values were estimated using the transfer function developed by Stoorvogel and Smaling (1990); in which the mean annual rainfall was the main source of data. The coefficients of 0.14, 0.023 and 0.092 were used for N, P and K respectively. Other scholars (Mulualem et al., 2021; Cobo et al., 2010; Lesschen et al., 2007; Van Beek et al., 2016) also used the same approach to estimate the atmospheric deposition of nutrients in various spatial scales. The rainfall data of the nearby stations of the watershed were used for this purpose.

$$\mathbf{IN3\ (N) = 0.14*}\sqrt{\boldsymbol{P}}$$

$$\mathbf{IN3\ (P) = 0.023*}\sqrt{\boldsymbol{P}}$$

$$\mathbf{IN3\ (K) = 0.092*}\sqrt{\boldsymbol{P}}$$

Where, the nutrient inflows are in kg/ha and p is rainfall in mm/yr.

Estimating the amount of nitrogen added to the system through N-fixation (IN4)

This is the forth path way of nutrient inflow (IN4) for nitrogen input. Free-living bacteria can fix atmospheric nitrogen (N_2) and fulfill 60% of the N requirement of legume crops (Haileslassie et al., 2005;Sheldrick et al., 2003). Smaling et al. (1993) also reported that 60% of the N uptake was fixed by legumes symbiotically. About 39% of the N requirement of plants can be fixed non-symbiotically at natural conditions (Dobereiner, 1996). In this study, the amount of nitrogen fixed (N4b) non-symbiotically was estimated using the transfer function (Lesschen et al., 2007; FAO, 2003).

$$\mathbf{IN4\ (N) = 0.5+\ 0.1*}\sqrt{\boldsymbol{P}}$$

Where, p is the amount of rainfall (mm/yr)

3.5.2.2. Quantification of nutrient outflows

Nutrient removal with crop harvest is the most important pathway of nutrient export from the agroecosystems. Two important parameters (productivity and nutrient contents) are required to know the amounts of nutrients removed with harvested products from the field. To quantify OUT1, yield data were collected from 9 farm fields using quadrants (4 m^2). These data were substantiated by the data on the average productivity of crops in the district. The nutrient contents of products were adopted from literature (FAO, 2003; Henao and Baanante, 2006)

(Table 15). Then, nutrients removed with crop harvest from the field (OUT1) were quantified using the method suggested by Gallagher et al. (2012).

OUT1 (N) = Y* N

OUT1 (P) = Y* P

Where, Y= yield (kg); N = nitrogen content (kg N/ ton of yield) and P= phosphorus content (kg P/ ton of yield).

Estimate nutrient exported with residues (OUT2)

Application of organic inputs in to the soil (leaving the residues on the field after harvest) will enhance SOM content and improve the availability of essential nutrients to crops. However, literature showed that around 80% of crop residue has been removed from the field for different purposes and substantial amounts of nutrients are exported from the agroecosystem (Elias et al., 1998; Kiros et al., 2014; Haileslassie et al., 2005). In this study, data on residue were collected through field measurements and the nutrient contents were obtained from the literature (FAO, 2003; Henao and Baanante, 2006). Finally, the nutrients exported with crop residues were estimated by adopting the nutrient contents of each crop from literature (Table 3.13) and using the following equations.

OUT2 (N) = R*N* F

OUT2 (P_2O_5) = R*P* F

Where, R is the amount of residue (kg/ha); F, is the residue removal factor; N, is nitrogen content (kg N/kg of residue); P, is phosphorus content (kg P/kg of residue).

Table 3.13. The amount of nutrient exported with crop harvest (yield and residue)

Crop type	N (kg/ton)		P (kg/ton)		K (kg/ton)		source
	yield	Residue	yield	Residue	yield	Residue	
Wheat	22.3	4.3	4.3	1.8	5.8	26.7	FAO=2003; Stoorvogel and Smaling (1990); Henao and Baanante (2006)
Teff	17.3	6.1	3.4	1.89	4.8	16.86	
Barly	15.5	7.0	2.8	1.6	6	21	
Maize	15.8	7 .6-11.8	9.4	3.0-5.8	5.7	23.0-28.4	

Nutrient loss through leaching (OUT3)

Easily soluble nutrients, like nitrogen and potassium, are prone to leaching loss from the surface of the soil. On the other hand, the literature showed that the loss of phosphorus by this process is almost null, since it is attracted and tightly held by soil particles (Carmo et al., 2017; Ahmed et al., 2006). Hence, leaching loss for P was not accounted for in this study. The degree of leaching varies depending on the amount of rainfall, soil characteristics and management strategies. In the present study, the amount of N and K losses through leaching were estimated using transfer functions described by Stoorvogel and Smaling (1990). They estimated these losses by considering soil fertility classes, annual rain fall and plants nutrient uptake of N and K. Other scholars (Mulualem et al., 2021; Cobo et al., 2010; Van Beek et al., 2016) also used the same approach in their study.

$$OUT3 (N) = 2.3 + (0.0021 + 0.0007 * F) * p + 0.3 * (IN1 + IN2) – 0.1 * TNU$$

$$OUT3 (K) = (0.6 + (0.0011 + 0.002 * F) * p+ 0.5 * (IN1 + IN2) – 0.1 * TKU)$$

Where P is the mean annual rainfall (mm); F: soil fertility class (1 = low, 2 = medium and 3 = high); IN1 + IN2: are mineral fertilizer and organic inputs applied (kg N and K h^{-1} $year^{-1}$) and TNU and TKU: total N and K uptake (kg h^{-1} $year^{-1}$) respectively.

Loss of N in the form of gas emission from farming systems (OUT4)

Denitrification and volatilization are the two processes of gaseous losses of nitrogen from the farming systems. Denitrification is a process that takes place in anaerobic conditions in which the soil is becoming entirely saturated. Loss of N through volatilization is linked with the amount of chemical and organic inputs applied. In the denitrification process, the loss of N is in the form of N_2O, NO and N_2; in the volatilization process, it is lost in the form of NH_3 (Stoorvogel and Smaling, 1990). To quantify N loss, the two processes are considered in combination (Mulualem et al., 2021; FAO, 2003).The loss of N through these processes was estimated using the regression equation developed by FAO (2003).

$$\mathbf{OUT4 (N) = (0.025 + 0.000855* P +0.01725 * IN1 + IN2 + 0.117* OC) + (0.113 * IN1 + IN2)}$$

Where P is the mean annual rainfall (mm/year); OC, soil organic carbon (%); IN1 + IN2 are mineral and organic fertilizers applied (kg N/ha/yr).

Estimating nutrient loss by soil erosion (OUT5)

Soil erosion was recognized as the major cause of soil nutrient depletion in the highlands of Ethiopia (Van Beek et al., 2016; Haileslassie et al., 2005). The nutrient loss due to soil erosion can be estimated using the rate of soil loss and the nutrient content of the sediment. Revised Universal Soil Loss Equation (RUSLE) was applied to estimate soil loss and sediment yield by water erosion. Five data inputs (rainfall erosivity, soil erodibility, slope length and steepness, land use land cover and management factors) were used to run the RUSLE model. Slope length and slope steepness factor was derived from the digital elevation model (DEM) of the study area.

$$A = R*K*LS*C*P$$

Where, A is annual soil loss rate (t/ha/yr); R is rainfall erosivity; K is soil erodibility; LS is, slope factor; C is cover factor, and P is management factor.

The contents of N, P and K exported with sediment were quantified by multiplying the amount of soil loss, nutrient contents and enrichment factors of N, P and K (Roy et al., 2003) (Table 3.14). Enrichment factors of 1.5 for N and K and 2 for P were used by Gachene (1995). Haileslassie et al. (2005) also used an enrichment factor of 1.5 for N, P and K; while Stoorvogel and Smaling (1990) used enrichment factor of 2 for all nutrients. In this study, an enrichment factor of 2.3, 2.8 and 3.2 proposed by Smaling et al. (2013); Wasige (2013) were used for N, P and K respectively.

N loss (kg/ha/yr) = enrichment factor * N content * soil loss (kg/hr/yr)

P loss (kg/ha/yr) = enrichment factor * P content * soil loss (kg/hr/yr)

K loss (kg/ha/yr) = enrichment factor * K content * soil loss (kg/hr/yr)

Table 3.14. The amount of nutrients exported with sediment (%) at different fertility levels

Soil fertility category	N content	P2O5 content	K2O content
1	0.05	0.02	0.05
2	0.1	0.05	0.1
3	0.2	0.1	0.2

1 = low soil fertility; 2 = medium soil fertility and 3 = high soil fertility. Source: Stoorvogel and Smaling (1990).

3.5.3. Nutrient balance calculation

The balances of N, P and K were calculated from the above described four nutrient inflows and five out flows. Both partial and full nutrient balance methods were applied. In partial nutrient balance calculation, only two nutrient inflows (IN1 and IN2) and two outflows (OUT1 and OUT2) were considered; whereas in full nutrient balance calculation, the four nutrient inflows and the five outflows were considered. Partial balance is more accurate than full balance as the flows are easily measured; however, as the name implies, they are incomplete since other nutrient fluxes are missed. Even though there are uncertainties in the full nutrient balances, they indicate the rates of soil nutrient losses (Smaling et al., 2013).

PNB = (IN1+IN2) - (OUT1+OUT2)

FNB = (IN1+IN2+IN3+IN4) - (OUT1+OUT2+OUT3+OUT4+OUT5)

Where, PNB = Partial nutrient balance and FNB = Full nutrient balance

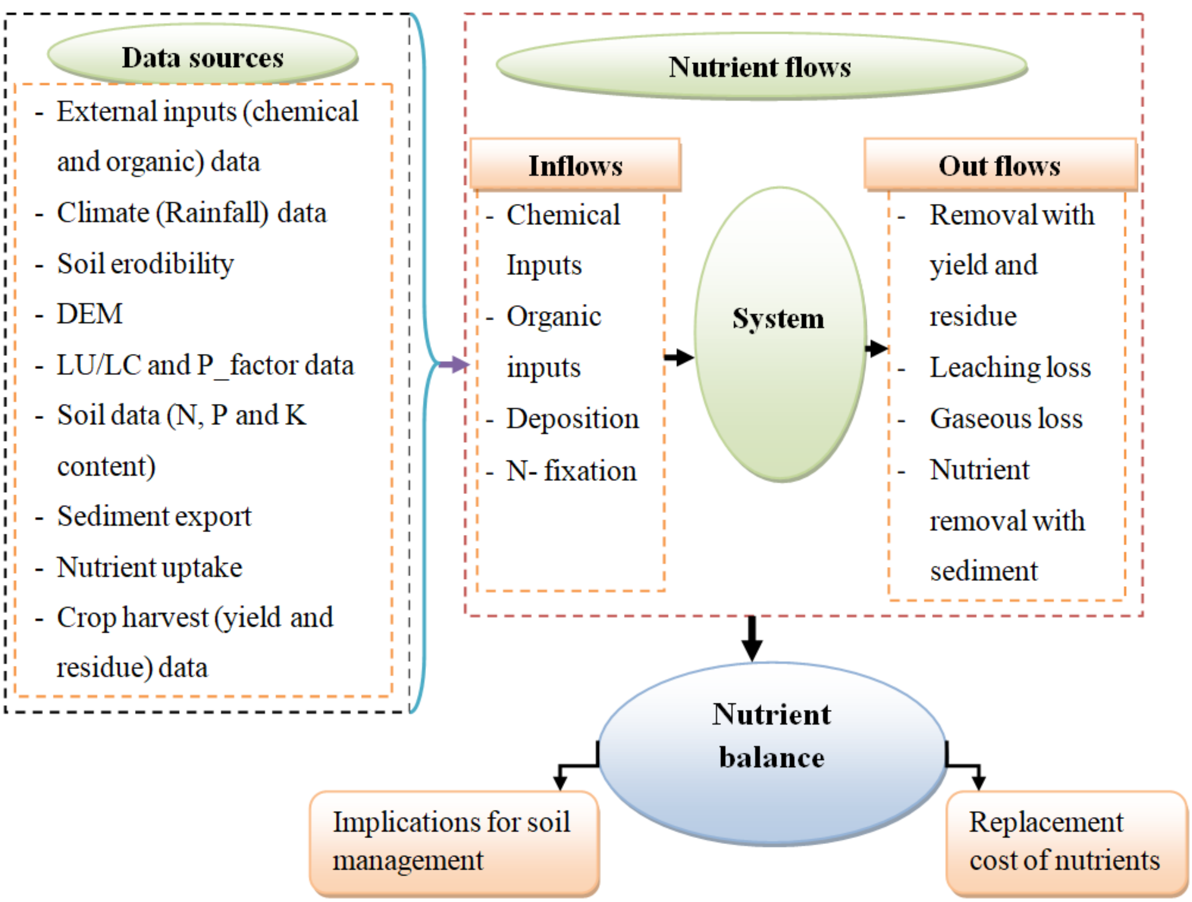


Fig 3.13. General frame work of the methodology

3.6. Data analysis

ENVI 5.3, ArcGIS 10.4 and Microsoft excel 2007 were used to analyze Landsat images, generate data and preparation of map layouts. Zonal statistics were also conducted to generate the value of LST in each land use/ land cover class. Correlation analysis was also employed to evaluate the relationship between NDVI and LST. Arc GIS and RUSLE models were also applied to estimate soil loss and sediment yield; identify vulnerable landscapes that demand immediate action. Two-way analysis of variance (ANOVA) following the generalized linear model (GLM) procedure was applied to test the difference between the mean values of soil quality indicators in different land use systems, elevation gradient and their interaction effects. The mean values of statistically different soil quality indicators were separated by Duncan's Multiple Range Test using SAS software version 9.0. In addition, linear correlation analysis was performed to explore the relationships among soil properties. Furthermore, principal component analysis (PCA) was applied as the method of factor extraction to identify key soil quality indicators (SQIs) and to group analyzed soil properties into major varimax rotated components of the soil quality index (SQI). The study area and soil maps were prepared using ArcGIS software version 9.5. Mean separations were also performed using least significant difference test ($p<0.05$).

CHAPTER FOUR: RESULTS

4.1. Detection of LULC and land surface temperature changes using GIS and remote sensing

4.1.1. Accuracy of Classified Images

The results of land use and land cover classification are summarized and presented below (Fig. 4.1 and Table 4.1). From the output maps of classified images, six types of land use and land cover categories were generated, including agricultural land, grazing land, forest land, shrub land, bare land, and built-up areas. In all study periods, the largest share of the watershed is covered by agricultural land, and the lowest proportion is forest land (1985 - 2019). The area covered by agricultural land increased dramatically from 44313.9 ha (55.2%) in 1985 to 59731.5 ha (74.4%) in 2019. In contrast, forest land covers 668.0 ha (0.8%) and 1076.5 ha (1.3%) in 1985 and 2019, respectively (Fig 4.2). Although it covers the lowest proportion, it has an increasing trend after 1999. In all the study periods agricultural land, bare land, and built-up area showed significant increments, while grazing land and shrub land decreased considerably.

Unless accuracy assessment has been carried out for classified images, errors will occur on the output maps. For this purpose, a confusion matrix was applied and results showed that the overall accuracies for classified images for 1985, 1999, 2009, and 2019 were 90.6%, 94.8%, 96.9%, and 95.9% respectively. In addition, the Kappa coefficient values were 0.86, 0.91, 0.89 and 0.87 for 1985, 1999, 2009, and 2019 (Table 4.2). The overall accuracy values are above the recommended minimum value of 85% suggested by (Eniolorunda and Bello, 2016) and the values of the kappa coefficient are more than 0.85, which shows the best agreement between classified images and referenced data (Monserud and Rick, 1992). Hence, the accuracy assessment results confirmed that further analysis can be done.

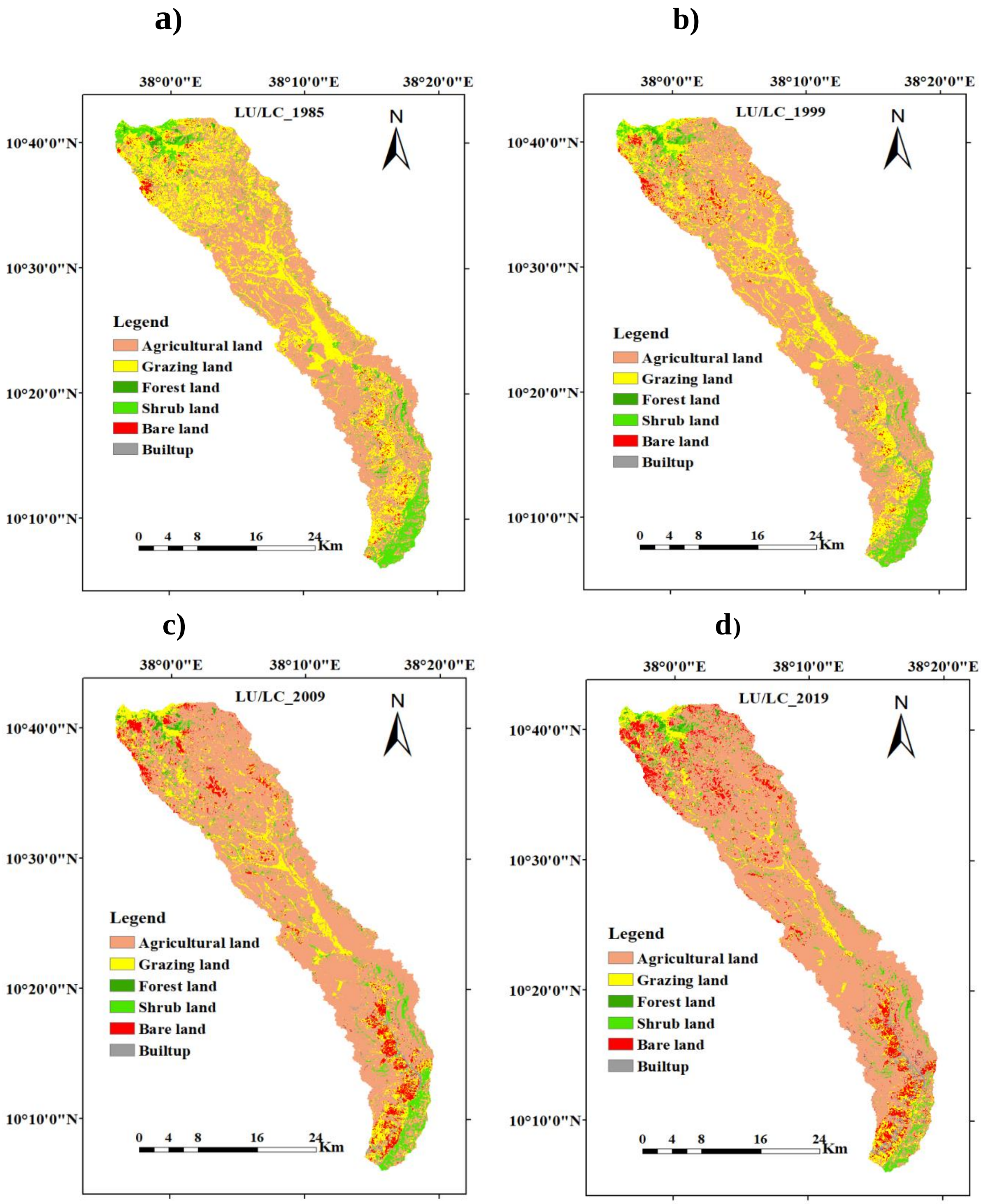


Fig. 4.1. LU/LC class maps of the Suha watershed for the year a) 1985; b) 1999; c) 2009, and d) 2019

Table 4.1. Area coverage of each land cover class in the Suha Watershed (1985- 2019)

LU/LC class	1985		1999		2009		2019	
	Area		Area		Area		Area	
	ha	%	ha	%	ha	%	ha	%
Agricultural land	44313.9	55.2	51938.9	64.7	57523.8	71.6	59731.5	74.4
Grazing land	25762.2	32.1	18272.3	22.7	11329.5	14.1	7193.8	9.1
Forest land	668.0	0.8	543.6	0.7	883.2	1.1	1076.5	1.3
Shrub land	7441.3	9.3	6592.5	8.2	4364.5	5.4	3897.1	4.9
Bare land	1417.7	1.8	1967.4	2.4	4854.5	6.1	6714.9	8.4
Built up	725.7	0.8	1016.9	1.3	1376.4	1.7	1719.7	2.0
Total	80328.8	100	80331.6	100	80331.9	100	80333.5	100

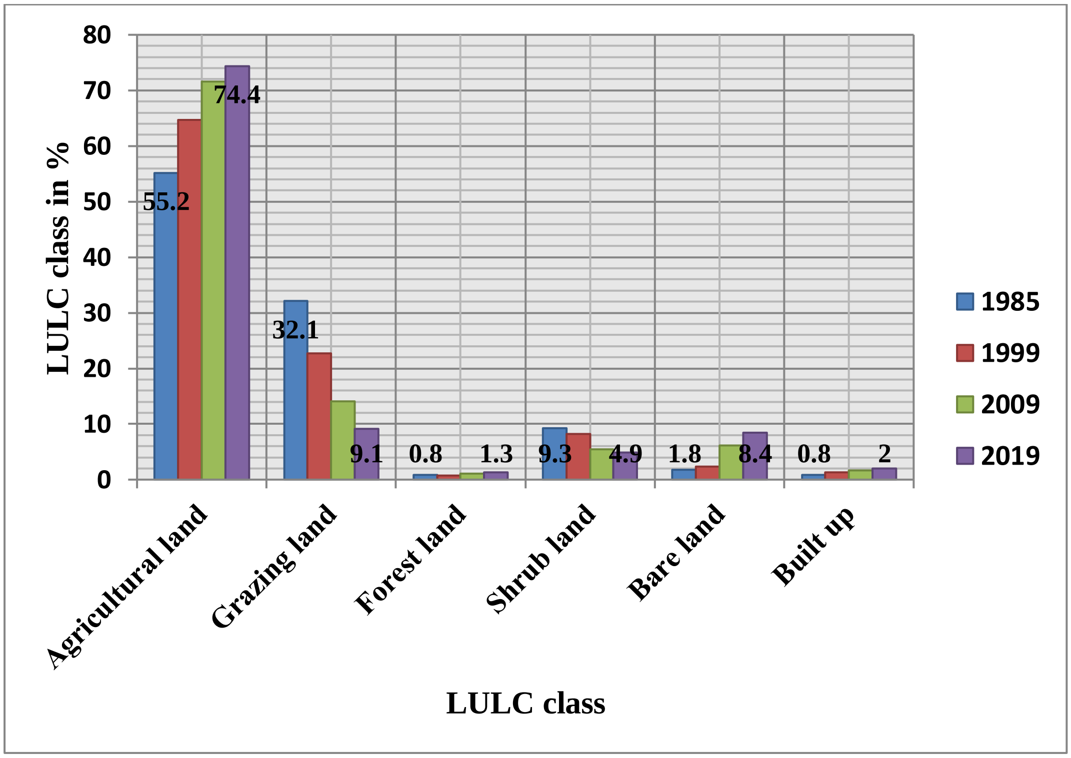


Fig. 4.2. Proportion (%) of LU/LC from the total area in suha watershed for consecutive study periods (1985 - 2019).

Table 4.2. Results of the accuracy evaluation of classified images (1985 – 2019)

LU/LC Class	1985		1999		2009		2019	
	UA (%)	PA (%)	UA (%)	PA (%)	UA (%)	PA (%)	UA (%)	PA (%)
Agricultural land	94.97	91.22	96.51	98.54	98.47	98.9	99.32	97.40
Grazing land	96.07	95.60	97.48	89.64	97.19	91.39	75.65	90.80
Forest land	81.28	84.83	86.89	95.17	99.53	82.56	95.12	80.43
Shrub land	78.67	84.22	89.56	82.97	49.63	68.56	77.11	91.04
Bare land	98.56	97.86	89.47	100	99.34	99.34	98.91	95.77
Built up	73.68	100	97.44	56.72	94.74	100	100	96.23
Over all accuracy (%)	**90.6%**		**94.8%**		**96.9%**		**95.9%**	
Kappa coefficient.	**0.86**		**0.91**		**0.89**		**0.87**	

Note: UA = User's accuracy; PA = Producer's accuracy

4.1.2. Change detection

A significant transition of LU/LC from one class to another was observed for the entire study period. The transitions of land-use types were evaluated by considering two consecutive periods, 1985 - 1999, 1999 - 2009, 2009 - 2019, and the entire period (1985 - 2019). Between 1985 and 1999, agricultural land, bare land, and built-up area were increased by 7625 ha (17.2%), 547.9 ha (38.8%), and 291.2 ha (40.1%), respectively. On the contrary, grazing land, forest land, and shrub land were reduced by 7489.9 ha (29.1%), 114.4 ha (17.1%), and 848.8 ha (11.4%), respectively (Table 4.3 and 4.4). In the second time interval (1999 - 2009), agricultural land, forest land, bare land, and built-up area were increased by 5584.9 ha (10.8%), 339.6 (62.5%), 2887.1 ha (146.7%), and 359.5 ha (35.4%) respectively. The forest land changed from decreasing trend to an increasing trend in this period. Grazing and shrub land were decreased by 6942.8 ha (38.0%) and 2228 ha (33.8%) respectively. The trend of land use/ land cover change is illustrated in (Fig 4.3). From the results of the transition matrix table, it is possible to detect the gains and losses of each land use and land cover class (Table 4.5). Values found on the diagonal of the matrix indicate persistent change, whereas off-diagonal values indicate conversion from one class to the other.

From 2009 to 2019, still, considerable changes have been observed following the same trend as in previous periods. Agricultural land, forest land, bare land, and built-up area were increased by 2207.7 ha (3.8%), 193.3 ha (21.9%), 1860.4 ha (38.3%), and 343.3 ha (24.9%), respectively. On the contrary, grazing and shrub land decreased by 4135.7 ha (36.5%) and 467.4 ha (10.7%), respectively. For the entire period (1985 -2019), highly significant transitions occurred. Agricultural land was greatly expanded (15417.6 ha) followed by bare land (5297.2 ha) and built area and forest land increased by 994 ha and 408.5 ha, respectively. However, grazing and shrub lands were reduced by 18568.4 ha and 3544.2 ha and converted to other types of land use.

Table 4.3. LULC changes between two consecutive periods in the Suha watershed from 1985-2019

Land use/cover type	LU /LC change (%)				Rate of change (ha/yr)			
	1985 -1999	1999-2009	2009-2019	1985-2019	1985 -1999	1999-2009	2009-2019	1985-2019
Agricultural land	17.2	10.8	3.8	34.8	508.3	558.5	220.8	453.5
Grazing land	-29.1	-38.0	-36.5	-72.1	-499.3	-694.3	-413.6	-546.1
Forest land	-18.6	62.5	21.9	61.2	-8.3	34.0	19.3	12.0
Shrub land	-11.4	-33.8	-10.7	-47.6	-56.6	-222.8	-46.7	-104.2
Bare land	38.8	146.7	38.3	373.6	36.6	288.7	186.0	155.8
Built up	40.1	35.4	24.9	137.0	19.4	36.0	34.3	29.2

Table 4.4. Net change (ha) of LU/LC in the Suha watershed from 1985 - 2019

Land use/cover type	Net change (ha)			
	1985 -1999	1999-2009	2009-2019	1985-2019
Agricultural land	7625	5584.9	2207.7	15417.6
Grazing land	-7489.9	-6942.8	-4135.7	-18568.4
Forest land	-124.4	339.6	193.3	408.5
Shrub land	-848.8	-2228	-467.4	-3544.2
Bare land	549.7	2887.1	1860.4	5297.2
Built up	291.2	359.5	343.3	994

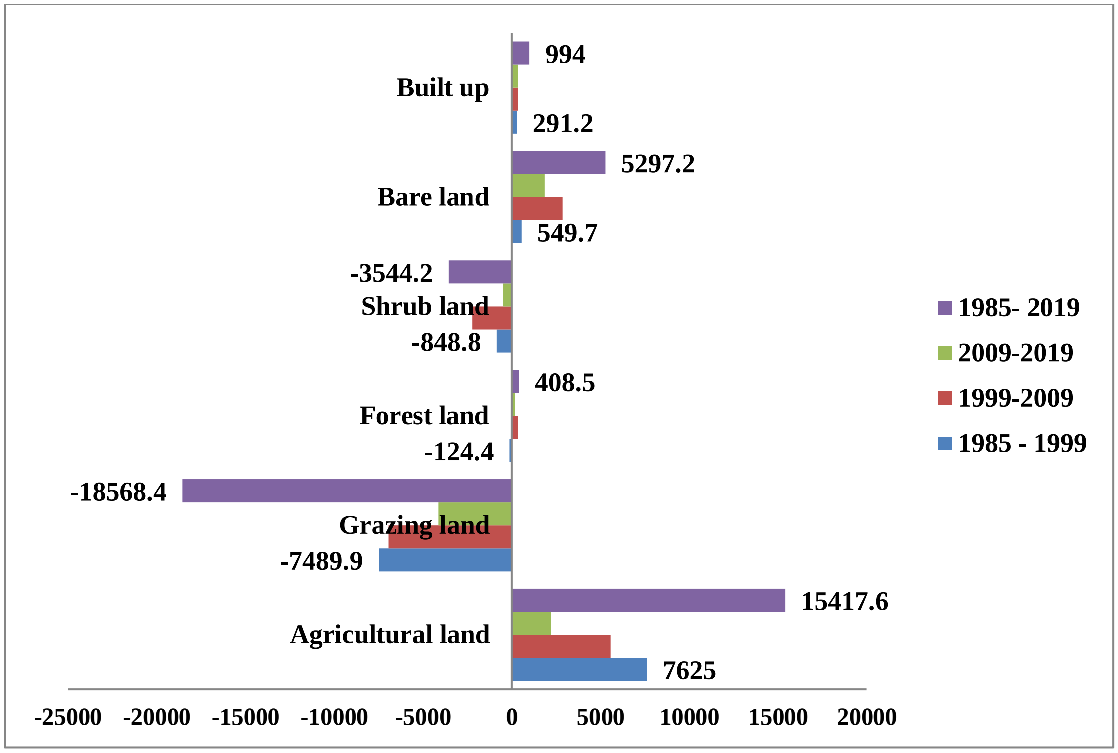


Fig. 4.3. Net change (ha) of the LULC classes in the Suha watershed for consecutive study periods (1985 -1999, 1999-2009, 2009-2019 and 1985-2019)

Table 4.5. Transition matrix (area in ha) between 1985 and 1999 in Suha watershed

1985	1999						
	Agricultural land	Grazing land	Forest land	Shrub land	Bare land	Built up	Grand total
Agricultural land	**38150**	3785	77	1653	146	486	44297
Grazing land	10663	**12921**	26	477	1315	347	25749
Forest land	91	14	**298**	264	0	1	668
Shrub land	2227	878	142	**4168**	9	11	7435
Bare land	390	549	0	21	**399**	58	1417
Built up	381	135	0	3	72	**134**	725
Grand total	51902	18281	543	6587	1941	1036	80291

Table 4.5 continued. Transition matrix (area in ha) between 1999 and 2009 in Suha watershed

1999	2009						
	Agricultural land	Grazing land	Forest land	Shrub land	Bare land	Built up	Grand total
Agricultural land	**47295**	1997	137	1316	688	478	51911
Grazing land	6774	**8207**	26	204	2680	390	18281
Forest land	83	14	**372**	72	0	1	543
Shrub land	2284	904	347	**2766**	218	67	6586
Bare land	614	122	0	1	**1153**	52	1942
Built up	454	81	0	2	113	**387**	1037
Grand total	57503	11325	883	4362	4851	1375	80300

Table 4.5 continued. Transition matrix (area in ha) between 1999 and 2009 in Suha watershed

2009	2019						
	Agricultural land	Grazing land	Forest land	Shrub land	Bare land	Built up	Grand total
Agricultural land	**52562**	857	299	1574	1794	421	57508
Grazing land	4156	**5201**	32	227	1389	319	11325
Forest land	97	35	**498**	253	0	0	883
Shrub land	1941	296	229	**1860**	25	11	4363
Bare land	609	723	0	2	**3226**	292	4852
Built up	396	123	0	1	231	**624**	1376
Grand total	59761	7235	1059	3916	6667	1668	80306

Table 4.5 continued. Transition matrix (area in ha) between 1985 and 2019 in the Suha watershed

1985	2019						
	Agricultural land	Grazing land	Forest land	Shrub land	Bare land	Built up	Grand total
Agricultural land	**39940**	1099	262	1065	1024	906	44296
Grazing land	15482	**4667**	134	459	4467	540	25749
Forest land	117	29	**364**	151	2	4	668
Shrub land	3308	1308	298	**2235**	228	57	7434
Bare land	457	123	1	3	**782**	51	1417
Built up	443	7	1	1	163	**110**	725
Grand total	59747	7234	1058	3915	6667	1668	80290

The diagonal values (bold values) in the matrix showed land use/ land cover classes that remained unchanged (persistent to change).

4.1.3. Spatial and temporal changes of LST in the Suha Watershed

The temporal and spatial variability of the land surface temperature for the study periods (1985 - 2019) is illustrated in Fig. 4.4 (a-d)). Spatial variability is observed from the output maps of LST. The highest value of LST was observed predominantly in the lower part of the watershed and the lowest value was observed primarily in the upper part of the watershed in all study periods. The summarized results of LST are also shown in (Table 4.6). In the first study period (1985), the LST value ranged from 10.4 ^{o}C to 40.7 ^{o}C with a mean value of 25.5 ^{o}C. In the year 1999, its value ranged from 9.5 ^{o}C to 41.8 ^{o}C with a mean value of 25.65 ^{o}C and in the last period (2019) its value ranged from 12.5 ^{o}C to 42.9 ^{o}C with the mean value of 27.7 ^{O}C. The mean value of LST has increased by 2.15 ^{o}C in 35 years.

4.1.4. Impacts of land use/cover change on land surface temperature in the suha watershed

Zonal statistical analysis was performed using the spatial analyst tool in ArcGIS10.4 to generate LST values in different land use land cover classes for each study period (Fig.4.5). Obtained results depicted that LST values (minimum, maximum and mean) vary significantly in different land use/ land cover classes and different locations. The mean value of LST ranged from 20.3 to 30.6 ^{O}C, 21.9 to 32.3 ^{O}C, 24.87 to 34.87 ^{O}C and 23.5 to 33.1 ^{O}C for the year 1985, 1999, 2009 and 2019 study periods, respectively. The highest value was recorded in built areas and bare land and the lowest value was recorded in forest land followed by shrub land in all study periods. From 1985 to 2019, the mean LST value of built area increased from 30.6 ^{O}C to 33 ^{O}C followed by bare land from 30.6 ^{O}C to 32.2 ^{O}C and in forest land, it increased from 20.3 ^{O}C to 23.5 ^{O}C (Table 4.6). From the results of the change in LU/LC, it is observed that impervious surfaces (bare land and built area) increased continuously over 35 years and these land uses increase LST throughout the study periods.

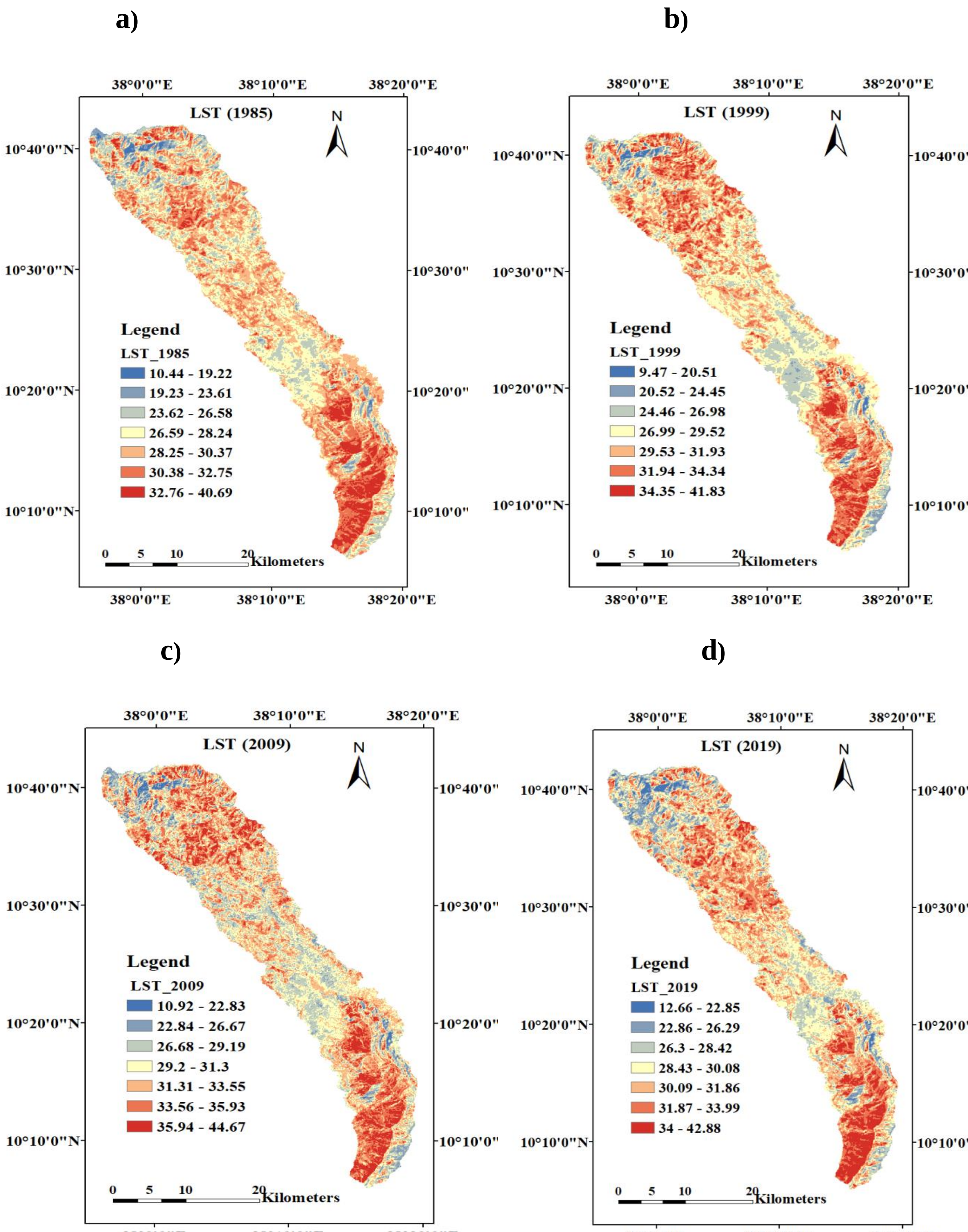


Fig.4.4. Land surface temperature maps of a) 1985; b) 1999; c) 2009 and d) 2019

Table 4.6. Mean values of land surface temperature (^oC) under different land use classes (1985 - 2019)

Land use class	1985 mean temperature (^oC)	1999 mean temperature (^oC)	2009 mean temperature (^oC)	2019 mean temperature (^oC)
Agricultural land	28.6	29.8	31.58	30.3
Grazing land	29.4	30.5	31.52	31.4
Forest land	20.3	21.9	23.93	23.5
Shrub land	26.7	26.9	27.87	27.0
Bare land	30.6	31.9	34.51	32.2
Built up	30.6	32.3	34.87	33.1

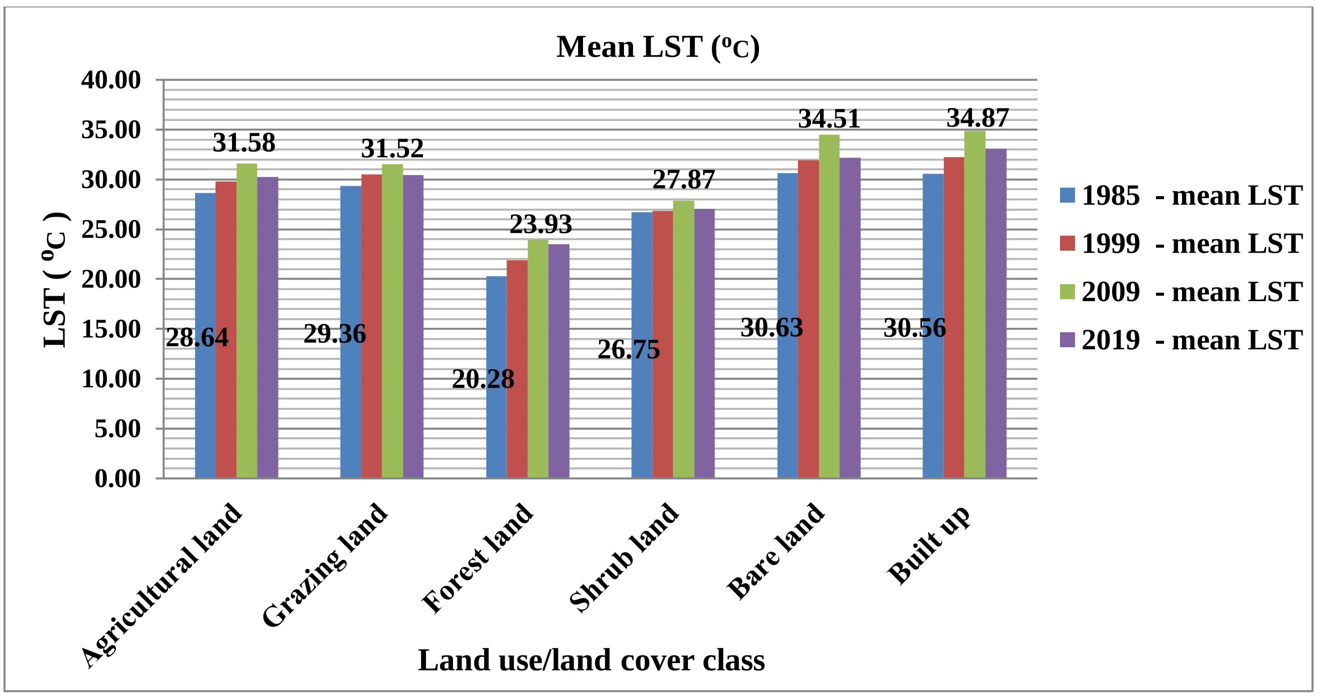


Fig. 4.5. Land use/cover change and LST relationship in the Suha watershed from 1985 to 2019

4.1.5. Relationship between NDVI and LST

To evaluate the relationships between LST and NDVI, a simple linear correlation was analyzed and the results are depicted in Fig8 (a-c). NDVI values are highly variable across locations in the watershed. The highest values of LST are recorded in land use /land cover classes that have the lowest NDVI value and vice versa (Fig 4.6 and Table 4.7). Areas covered with forests have the highest NDVI value but the lowest LST value. On the other hand, areas covered with built-up and bare land have the lowest NDVI value but the highest LST value.

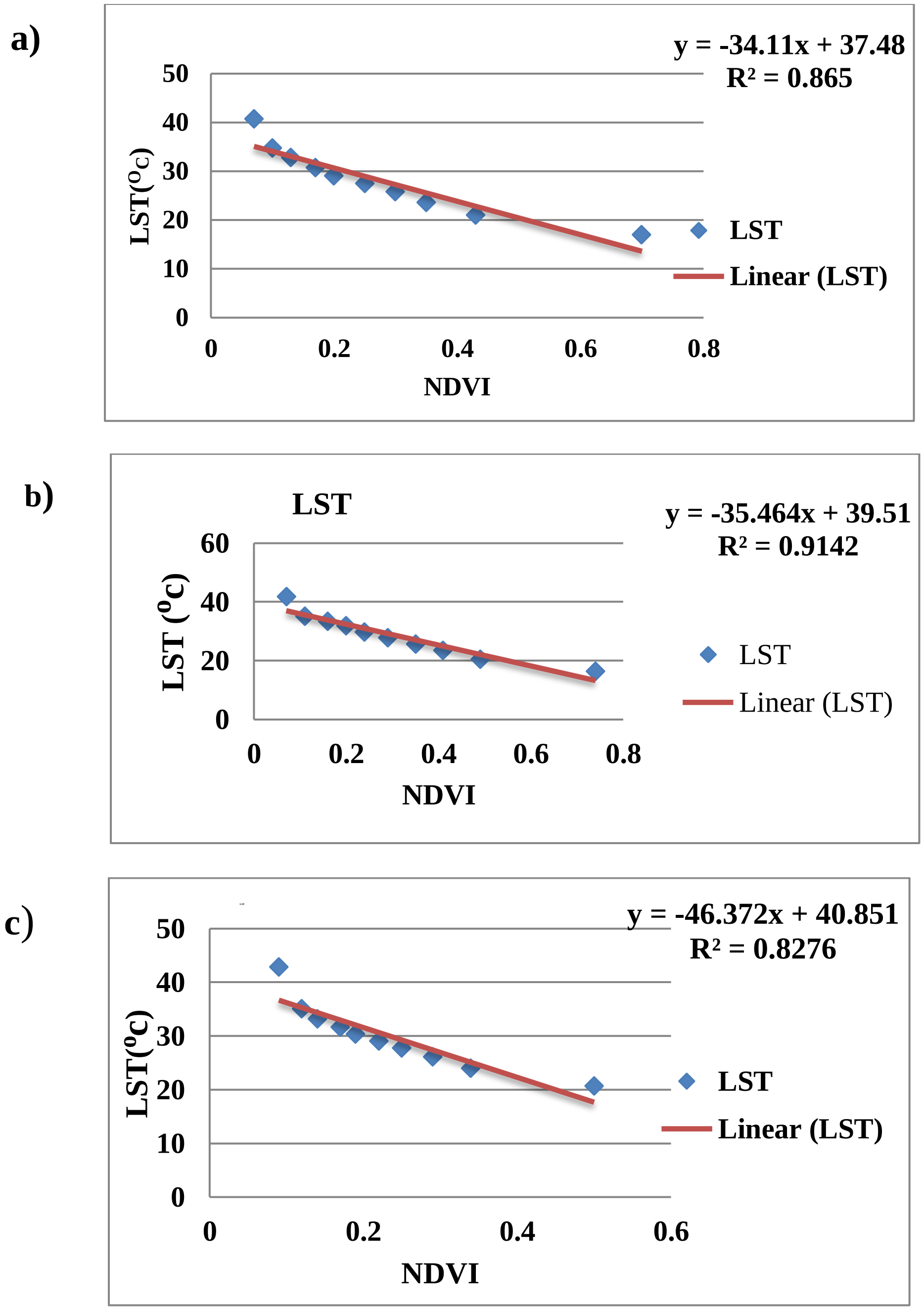


Fig. 4.6. Correlation between NDVI and LST in the Suha watershed for a) 1985; b) 1999 and c) 2019 periods.

Table 4.7. Coefficient of determination and coefficient of correlation of NDVI versus LST

period	NDVI versus LST	
	Coefficient of determination (R^2)	Coefficient of correlation (R)
1985	- 0.865	- 0.930
1999	- 0.914	- 0.956
2019	- 0.827	- 0.909

Note: The - sign indicates an inverse relationship between the two variables (NDVI and LST).

4.2. Quantification of soil erosion and sediment yield in response to land use land cover change using GIS and RUSLE model

Extent and spatial variability of soil loss rate in the Suha watershed (1985-2019)

The trend of annual soil loss, and its spatial distribution in the watershed are depicted in (Fig 4.7; Table 4.8 and 4.9). Total soil loss of the catchment was 1,221,214; 1,751,477; 2,522,770 and 2,426,359t/yr in 1985, 1999, 2009 and 2019 respectively. The highest value of annual soil loss was observed on the upper and lower part of the watershed where there are steep slope landscapes and poor land management strategies. The mean annual soil loss rates also showed temporal and spatial variability; 15.2, 21.8, 31.4 and 30.2 t/ha/yr for 1985, 1999, 2009 and 2019 respectively. This variability is also linked with gross annual soil loss.

Estimates of sediment export in the study area

The analyzed results of SDR indicated that its mean values were 0.26 using catchment area as data input (first approach) and 0.21using channel bed slopes as data input (second approach) in empirical equations mention in the methodology part. Sediment yield was estimated by integrating gross annual soil loss and SDR. Obtained results were 3.95, 5.66, 8.16 and 8.02 t/ha/yr for 1985, 1999, 2009 and 2019 respectively for the first approach and for the second approach; the values were 3.19, 4.57, 6.6 and 6.4t/ha/yr for the same period (Fig 4.8). Sediment deposition is the amount of sediment displaced from the upstream area and deposited on gentle slopes and depression areas within the catchment. The total annual sediment deposition values were 903,698; 1,296,093; 1,866,850 and 1,795,505t/yr for the study periods.

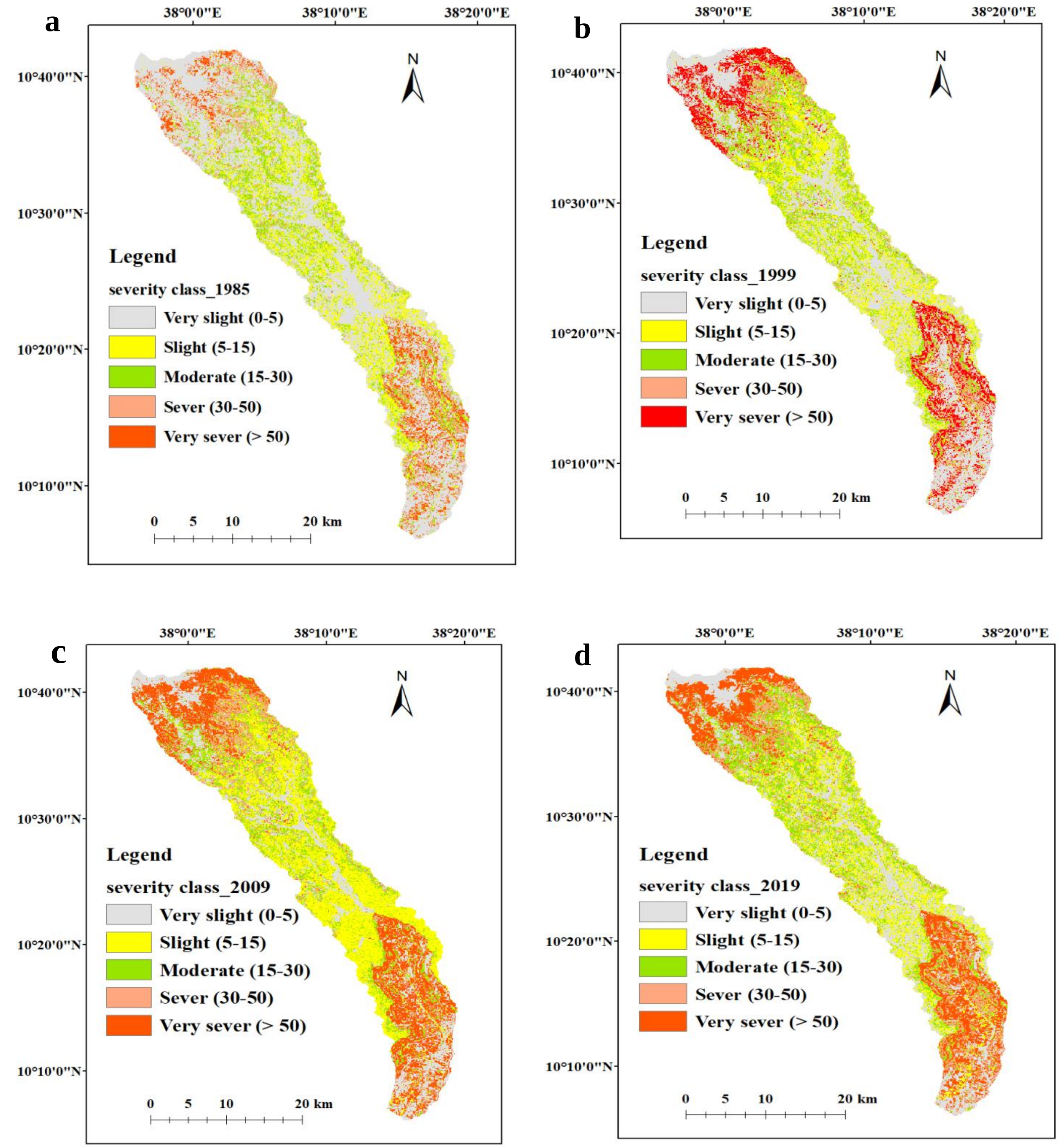


Fig 4.7. Spatio-temporal variability and severity class maps of soil erosion for a) 1985; b) 1999; c) 2009 and d) 2019 in the Suha Watershed

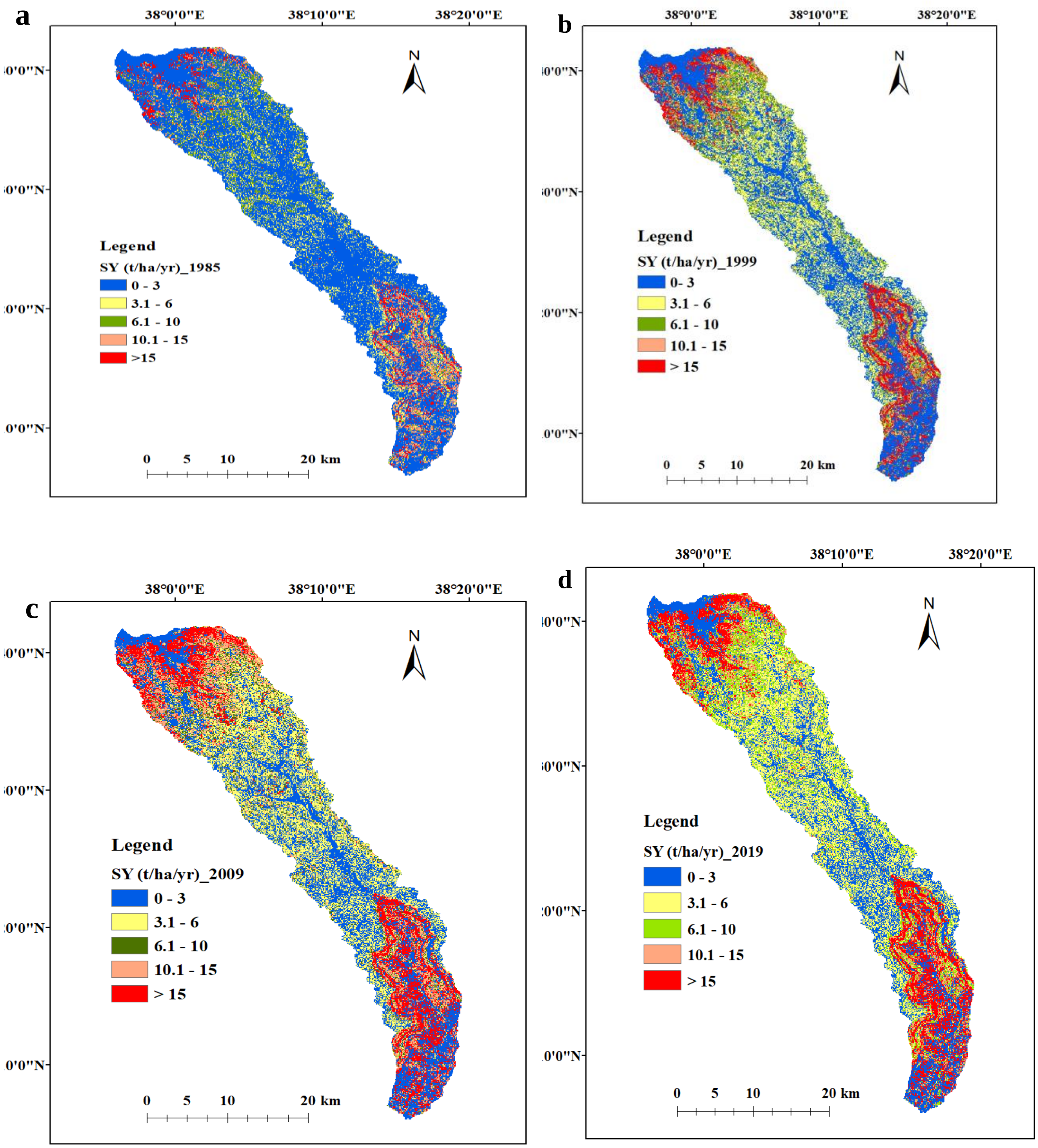


Fig 4.8. Spatio-temporal variability map of sediment export for a) 1985; b) 1999; c) 2009 and d) 2019 in Suha watershed

Table 4.8. Soil loss and sediment export trends of Suha watershed in the last 35 years.

year	Rate of soil loss (t/ha/yr)	Total annual soil loss (t/yr)	Mean sediment yield (t/ha/yr)	Total annual sediment yield (t/yr)
1985	15.2	1,221,214	3.95	317,515.5
1999	21.8	1,751,477	5.66	455,384.1
2009	31.4	2,522,770	8.16	655,920.3
2019	30.2	2,426,359	8.02	630,853.2

Table 4.9. Soil erosion severity classes and area coverage in Suha watershed (1985-2019)

Soil loss rate (t/ha/yr)	Severity class	1985		1999		2009		2019	
		Area (ha)	**(%)**	**Area (ha)**	**(%)**	**Area (ha)**	**(%)**	**Area (ha)**	**(%)**
0-5	Very slight	42957	53.5	32624	40.6	16223.9	20.2	22252	27.7
5-15	Slight	13826	17.2	14930	18.6	24881.6	31.0	16788	20.9
15-30	Moderate	10852	13.5	13350	16.6	12211.7	15.2	15643	19.5
30-50	Severe	6126	7.6	8020	10.0	8572.9	10.7	9138	11.4
>50	Very severe	6582	8.2	11419	14.2	18452.9	23.0	16522	20.6
	Total	**80343**	**100**	**80343**	**100**	**80343**	**100**	**80343**	**100**

Severity classification is based on Degife et al. (2021); Yesuph and Dagnew, (2019); Haregeweyn et al. (2017).

4.2.1. Impacts of LU/LC change on soil erosion in the Suha Watershed

Zonal statistics in Arc GIS spatial analyst tool was applied to see the impacts of land use land cover classes on soil erosion. Raster maps of LU/LC classes were used as the input raster and land cover class as the zonal field and soil loss map as the input value raster. Significant variation was observed in the values of mean annual soil loss rate among different land use/cover classes. The highest mean value of soil loss was observed in bare lands followed by cultivated land and built-up areas (Fig 4.9 and Table 4.10). The lowest value was recorded in forest land followed by shrub land. Slope length and slope steepness play a great role in the processes of soil erosion. Significant differences were also observed in soil erosion rate in different landscape positions of the watershed. Landscape positions with slope gradients greater than 30% showed a mean soil loss rate from 26.6 to 60.9t/ha/yr (Fig 4.10 and Table 4.11).

Table 4.10. Mean soil loss in different land use/ land cover classes in the Suha Watershed (1985-2019)

Land use type	Mean annual soil loss Rate (t/ha/yr)			
	1985	1999	2009	2019
Agricultural land	17.3	24.4	29.4	27.5
Grazing land	10.3	13.1	18.0	19.2
Forest land	6.6	10.4	14.0	13.4
Shrub land	12.8	16.6	18.5	20.4
Bare land	53.7	57.6	102.7	81.9
Built up	16.8	20.5	32.4	32.0
Total mean	15.2	21.8	31.8	30.4

Table 4.11. Mean annual soil loss Rate (t/ha/yr) in different slope classes in Suha watershed

Slope class (%)	Mean soil loss Rate (t/ha/yr)			
	1985	1999	2009	2019
0-5	6.4	8.3	10.2	10.1
5-10	8.6	11.7	14.7	14.6
10-20	14.1	20.9	28.6	28.8
20-30	22.4	32.3	50.1	49.4
30-50	27.8	38.0	60.9	58.6
>50	26.6	41.5	60.6	57.9
Over all mean (t/ha/yr)	15.2	21.8	31.4	30.2

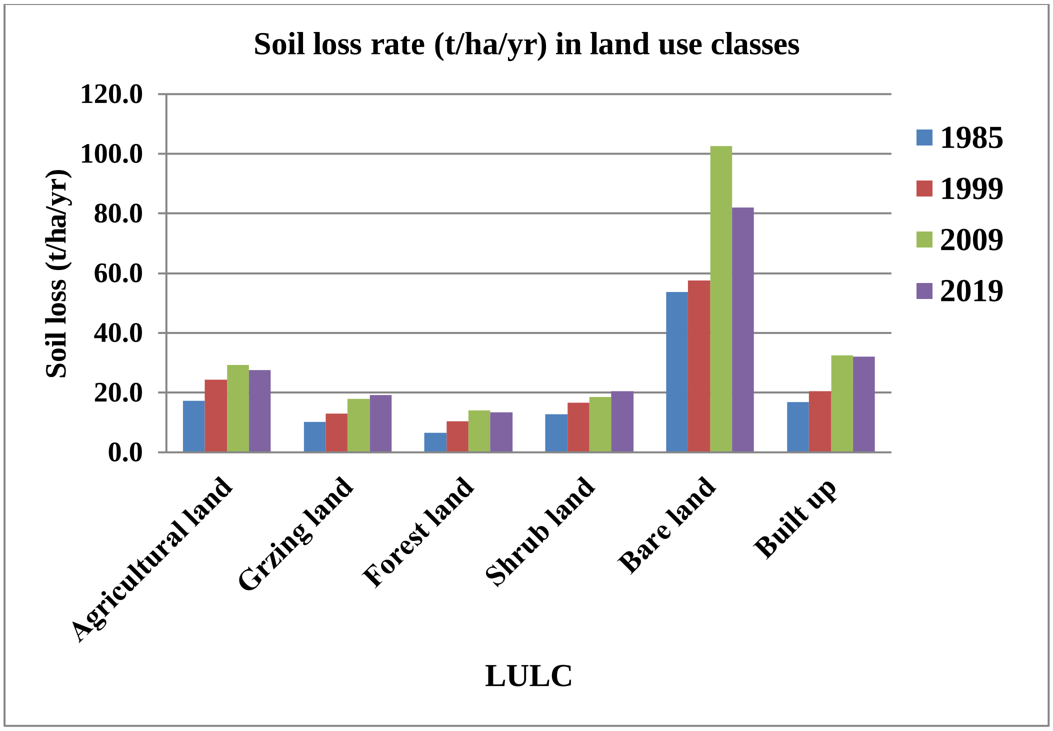


Fig 4.9. Variation in mean soil loss in d/t land use classes and periods

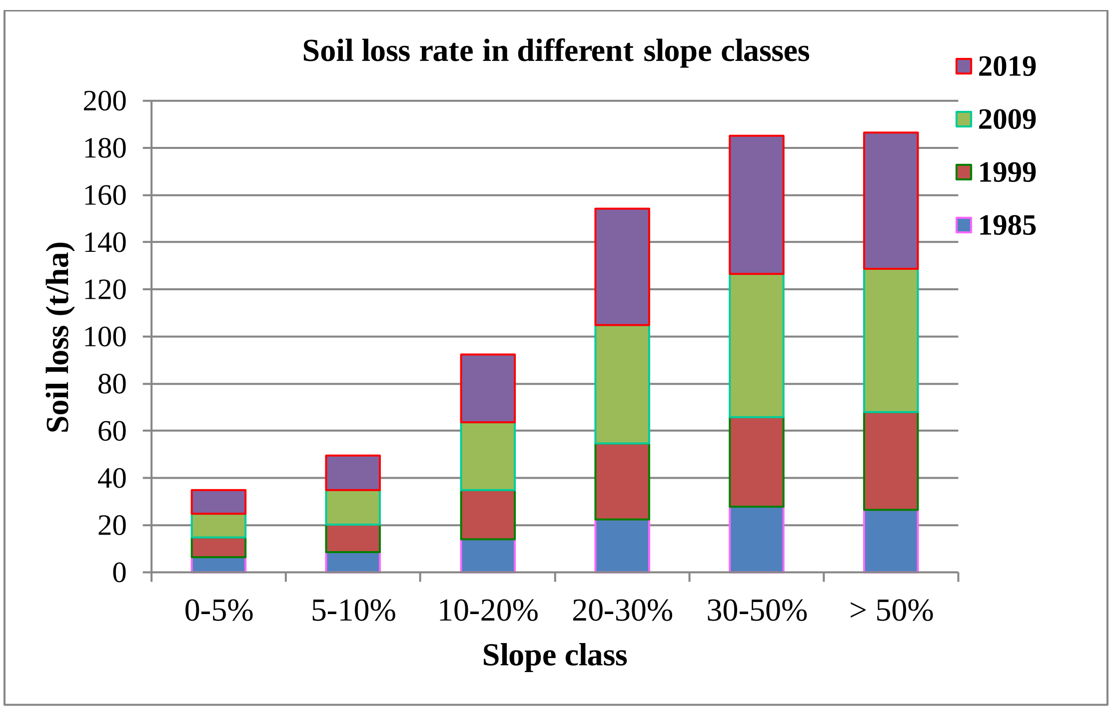


Fig 4.10. Soil loss rate (t/ha) in different slope class and periods

4.2.2. Soil erosion severity in sub watersheds (1985 - 2019)

Soil erosion severity classes were analyzed by reclassifying the soil loss map and adopting the class ranges of Haregeweyn et al. (2017). The analyzed results ranged from very slight to very severe severity class. In the entire watershed, significant variation was observed among sub-watersheds in terms of their spatial distribution of soil erosion risk (Fig 4.11 and Table 4.12). Fourteen sub-watersheds are classified under severe and very severe soil erosion severity classes and these sub-watersheds are found in the slope gradient greater than 30%. The results also showed that from the total area of the watershed 55.5% experienced greater than 30t/ha/yr mean annual soil loss.

Table 4.12. Mean soil loss and SY of sub-watersheds

No.	Sub watershed	Area (ha)	Rate of soil loss(t/ha/yr)				Mean SY (t/ha/yr)			
			1985	1999	2009	2019	1985	1999	2009	2019
1	Sw1	1901.1	16.3	28.1	46.8	46.3	4.2	7.3	12.2	12.0
2	Sw2	8017.6	19.0	32.9	48.2	43.2	4.9	8.5	12.5	11.2
3	Sw3	1803.6	19.8	38.1	39.8	43.0	5.1	9.9	10.4	11.2
4	Sw4	7049.5	15.1	26.9	37.1	39.3	3.9	7.0	9.7	10.2
5	Sw5	4942.9	10.7	16.2	20.3	20.1	2.8	4.2	5.3	5.2
6	Sw6	2112.9	9.2	11.4	14.4	14.3	2.4	3.0	3.7	3.7
7	Sw7	6033.9	10.1	16.6	21.1	21.5	2.6	4.3	5.5	5.6
8	Sw8	3366.4	11.0	13.3	16.3	17.1	2.9	3.4	4.2	4.4
9	Sw9	1617.4	9.3	11.2	13.2	13.1	2.4	2.9	3.4	3.4
10	Sw10	3217.2	8.5	11.0	13.6	14.3	2.2	2.9	3.5	3.7
11	Sw11	2912.6	6.3	9.3	12.3	12.6	1.6	2.4	3.2	3.3
12	Sw12	7195.7	8.1	11.1	13.2	12.7	2.1	2.9	3.4	3.3
13	Sw13	3436.7	17.4	19.9	41.7	38.0	4.5	5.2	10.8	9.9
14	Sw14	2568.3	22.1	26.3	44.4	47.3	5.7	6.8	11.5	12.3
15	Sw15	1810.9	20.1	18.5	51.6	56.4	5.2	4.8	13.4	14.7
16	Sw16	956.3	21.6	33.1	45.4	45.9	5.6	8.6	11.8	11.9
17	Sw17	4643.1	26.6	32.7	51.8	47.0	6.9	8.5	13.5	12.2
18	Sw18	827.5	27.4	33.8	69.5	64.6	7.1	8.8	18.1	16.8
19	Sw19	4075.4	15.2	21.8	26.2	25.5	3.9	5.7	6.8	6.6
20	Sw20	2374.6	23.8	33.0	40.2	41.8	6.2	8.6	10.5	10.9
21	Sw21	1402.2	18.9	23.9	40.6	40.6	4.9	6.2	10.6	10.6
22	Sw22	2523.8	18.5	22.6	32.6	30.0	4.8	5.9	8.5	7.8
23	Sw23	2953.7	25.6	33.2	46.6	46.7	6.7	8.6	12.1	12.2
24	Sw24	1882.1	6.6	8.9	11.1	11.0	1.7	2.3	2.9	2.9

SY= Sediment yield

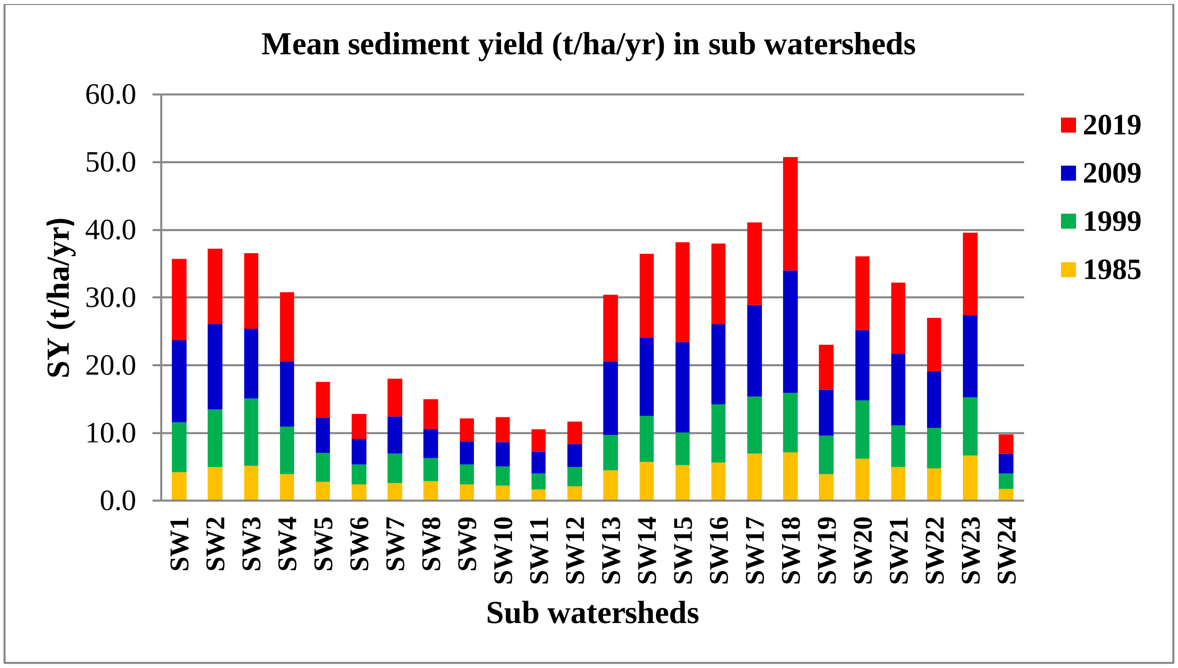


Fig 4.11. Mean sediment yield (SY) variability in sub watersheds during the study period (1985-2019).

Validation of RUSLE results

Validation of model results is required to see the fitness of the model and the accuracy of modeled results with measured values. However, the absence of measured sediment yield data in most catchments is the basic problem for this task. The same holds for the study area, in which measured sediment yield data are very fragmented and even in some years, data is missed for consecutive years. Due to this reason, analyzed results were validated using the findings of previous studies conducted in the north-western highlands of Ethiopia. Moreover, a field survey was conducted and related information was gathered from development agents and households. Similar approaches were used by Bekele and Gemi (2021); Fenta et al. (2021); Haregeweyn et al. (2017) due to lack of measured data for validation.

4.3. Monitoring soil quality in terms of its physical and chemical fertility along the toposequence of the watershed

Selected soil physical and chemical quality indicators were investigated under different land use systems and elevation gradients. The results of descriptive statistics illustrated the minimum and maximum values of soil quality indicators which revealed the existence of a high degree of variability in the study watershed (Table 4.15). The difference in the land use system/land management, elevation gradient, landscape position and soil type might be the factors for this variability. The results of ANOVA, correlation analysis and PCA are presented and discussed below.

Table 4.13. Descriptive statistics of analyzed soil physico-chemical properties in the Suha Watershed

Soil properties	**No. of samples**	**Minimum**	**Maximum**	**Mean**	**SE**	**CV (%)**
Physical properties						
Sand (%)	27	0.24	49.06	13.21	2.57	177.98
Clay (%)	27	5.66	86.04	53.72	3.73	375.72
Silt (%)	27	12.47	45.28	33.07	1.77	84.9
ISS	27	1.05	10.78	4.65	0.53	7.60
FC (%)	27	36.47	72.41	48.57	1.59	68.70
PWP (%)	27	25.12	49.28	33.98	1.11	33.41
AWHC (%)	27	10.00	23.13	14.59	0.57	8.73
Chemical properties						
pH-H_2O	27	4.93	6.31	5.78	0.06	0.11
OC (%)	27	0.38	4.60	2.27	0.23	1.44
TN (%)	27	0.05	0.35	0.15	0.01	0.00
C: N	27	7.6	20.14	15.11	0.72	13.83
Av.P (mg kg^{-1})	27	10.10	64.50	27.20	3.87	223.27
Ex. Na^+ (cmol(+) kg^{-1})	27	0.48	2.51	1.37	0.09	0.21
Ex. K^+ (cmol(+) kg^{-1})	27	0.17	3.28	1.07	0.13	0.49
Ex. Ca^{2+} (cmol(+) kg^{-1})	27	18.94	58.77	37.69	2.07	116.26
Ex. Mg^{2+} (cmol(+) kg^{-1})	27	7.36	22.14	13.59	0.78	16.74
CEC (cmol(+) kg^{-1})	27	45.58	88.55	69.44	2.24	135.31
BS (%)	27	56.9	99.4	76.48	2.24	148.51

4.3.1. Effects of land use systems and elevation gradient on selected soil physical quality indicators

Particle size distributions, Index of soil aggregate stability (ISS) and soil water content

The analysis of variance results of soil physical quality indicators under different land use types and elevation gradients were depicted in (Table 4.16**).** The results revealed that soil separate groups (sand, clay and clay) showed significant differences ($p < 0.05$) between agricultural and forest land use types in the upper part of the watershed. In the other cases, non-significant differences were observed in the main and interaction effects of land use systems and locations. The mean value of sand ranged from 2.55% to 35.12%; for clay fraction it ranged from 19.85% to 72.66% and for silt fraction it ranged from 24.78% to 47.36%. The textural class of soils in most locations and land use systems is clay; other textural classes include clay loam, silt clay and sand clay. When clay fraction is removed from the surface by soil erosion the content of sand increases which is evidenced from negative and significant correlation ($r = -0.755$, $p < 0.01$) between these fractions.

Significant differences ($p < 0.05$) were also observed for ISS among treatments (land use systems and elevation gradients) and their interaction effects. The lowest (2.06%) and highest (7.21%) values were recorded in the agricultural land and forest land use system respectively. Pieri (1992) classify structural degradation of soils as stable structure when ISS > 9%; low risk of structural degradation when 7% < ISS < 9%; high risk of structural degradation when 5% < ISS < 7% and structurally degraded soil when ISS < 5%. Based on these classifications, soils under agricultural and grazing lands are rated as structurally degraded soils where as soils under forest land are rated as low risk of structural degradation.

The mean value of soil water content at field capacity (FC) ranged from 36.94% to 59.08% in the clay loam textural class and from 40.31% to 72.41% for clay textural class. For the same textural classes, the values of permanent wilting point (PWP) ranged from 25.52% to 43.91% and from 28.84% to 49.28%. For available water content (AWC) the values ranged from 11.42% to 15.17%; and from 11.47% to 23.13% respectively. Statistically significant difference ($p < 0.05$) was observed for PWP between the mean values of agricultural land and forest land in the mid land of the watershed. Though significant differences were not observed for soil physical quality indicators, marked differences were observed as they are altered by land use systems and elevation gradients.

Table 4.14. Mean values of selected soil physical properties under different land use systems and elevation gradient.

Soil property	Elevation gradient	Land use system			Mean ±SE
		Cultivated land	Grazing land	Forest land	
Sand	upper	2.55b± 6.6	15.18ab± 6.6	35.12a± 6.6	17.62± 3.84
	middle	14.78a± 6.6	14.34a± 6.6	3.19a± 6.6	10.76± 3.84
	lower	15.45a± 6.6	7.62a± 6.6	10.65a± 6.6	11.24± 3.84
	Mean ±SE	10.93[a] ±3.84	12.38[a] ±3.84	16.32[a] ±3.84	13.21± 2.22
Clay	upper	72.66a± 9.4	49.94a± 9.4	19.85a± 9.4	47.48± 5.45
	middle	54.21a± 9.4	51.53a± 9.4	61.03a± 9.4	55.59± 5.45
	lower	56.53a± 9.4	56.52a± 9.4	61.20a± 9.4	58.08±5.45
	Mean ±SE	61.13[a] ±5.45	52.66[a] ±5.45	47.36[a] ±5.45	53.72 ± 3.15
Silt	upper	24.78b± 5.0	34.88ab± 5.0	45.03a± 5.0	34.89 ± 2.89
	middle	31.02a± 5.0	34.13a± 5.0	35.77a± 5.0	33.64± 2.89
	lower	28.03a± 5.0	35.86a ± 5.0	27.14a ± 5.0	30.68± 2.89
	Mean ±SE	27.94[a] ±2.89	34.96[a] ±2.89	36.31[a] ± 2.89	33.07 ± 1.67
ISS	upper	1.78 [c] ± 0.78	4.12 [b]± 0.78	10.29 [a]± 0.78	5.40[a]± 0.46
	middle	2.32[b] ± 0.78	4.95 [a] ± 0.78	5.42 [a] ± 0.78	4.23 [a] ± 0.46
	lower	2.09 [b] ± 0.78	4.95 [a] ± 0.78	5.92 [a] ± 0.78	4.32 [a] ± 0.46
	Mean ±SE	2.05[c]±0.46	4.67[b]±0.46	7.21[a]±0.46	4.65± 0.26
FC (v %)	upper	48.97a±4.36	50.71a ±4.36	41.76a ±4.36	47.14 ± 2.52
	middle	41.24a±4.36	46.62a ±4.36	48.63a ±4.36	45.49± 2.52
	lower	60.10a ±4.36	47.35a ±4.36	51.71a ±4.36	53.05± 2.52
	Mean ±SE	50.1[a] ±2.52	48.23[a] ±2.52	47.37[a] ±2.52	48.56 ± 1.45
PWP (v %)	upper	33.59a± 2.95	35.2a± 2.95	29.09a± 2.95	32.62 ± 1.71
	middle	28.31b± 2.95	32.57ab± 2.95	34.79a± 2.95	31.89± 1.71
	lower	42.21a± 2.95	34.04a± 2.95	36.00a± 2.95	37.41± 1.71
	Mean ±SE	34.71[a] ±1.71	33.94[a] ±1.71	33.29[a] ±1.71	33.98 ± 0.99
AWC(v %)	upper	15.39a±1.71	15.51a±1.71	12.67a±1.71	14.52 ± 0.99
	middle	12.92a±1.71	14.05a±1.71	13.84a±1.71	13.60± 0.99
	lower	17.89a±1.71	13.31a±1.71	15.71a±1.71	15.63± 0.99
	Mean ±SE	15.40[a]±0.99	14.29[a]±0.99	14.07[a]±0.99	14.59± 0.57

Means in the rows followed by the same letter are not significant

4.3.2. Effects of land use systems and elevation gradient on selected soil chemical quality indicators

Soil pH, organic carbon (OC), total nitrogen (TN), C: N and available phosphorus (Av.P)

The analysis of variance results revealed that the mean value of soil pH ranged from 5.56 to 6.13; which is classified as moderately acidic (EthioSIS, 2014). The lowest value (5.56) and the highest value (6.13) were found in the mid-altitude and low altitude of the cultivated fields. Non-significant variations were observed between land use systems, elevation gradient as well as their interaction effects in upper and mid altitudes; whereas, in the lower altitude significant variation was observed between the cultivated land and other land use systems (Table 4.17).

The mean value of soil organic carbon (SOC) ranged from 1.01% to 3.89%. In all elevation gradients, the lowest values were recorded in the agricultural land whereas the highest values were in the forest land (Fig 4.12). Significant differences were observed between the main treatments (land use types and locations) and their interaction effects (LU*L). Based on the ratings of EthioSIS (2014), the contents of SOC are very low in the agricultural lands in all locations, low in the grazing lands and moderate in the forest land use system. This indicates that soil quality; particularly soil biological indicators are highly degraded in the cultivated and grazing land use systems.

The results also showed that the mean values of soil TN ranged from 0.08% (agricultural land) to 0.21% (forest land) (Fig 4.13). Significant differences ($p < 0.05$) were observed in the main and interaction effects of land use systems and elevation gradients. A significant difference was observed between agricultural land and the other two land use systems. The status of TN is rated as very low in the agricultural land and moderate in the forest and grazing land use systems EthioSIS (2014). The mean values of the C: N ratio also ranged from 11.22 to 18.71 and the overall mean values of the C: N ratio were 12.98, 14. 77 and 17.56 in the cultivated land, grazing land and forest land respectively. Significant difference ($p < 0.05$) was observed between cultivated land and forest land. Though non-significant variations were observed, the numerical values were higher in the grazing land as compared to cultivated land. The same is true for forest land and grazing land (Fig 4.14).

Table 4.15. Mean values of pH, OC, TN, C: N and Av.P under different land use systems and elevation gradient.

Soil property	**Elevation gradient**	**Land use system**			**Mean ±SE**
		Cultivated land	Grazing land	Forest land	
pH-H_2O	Upper	5.85^{a}± 0.19	5.64^{a}± 0.19	5.99^{a}± 0.19	5.82 ± 0.11
	middle	5.56^{a}± 0.19	5.61^{a}± 0.19	5.83^{a}± 0.19	5.67± 0.11
	lower	6.13a± 0.19	5.62b± 0.19	5.78b± 0.19	5.84± 0.11
	Mean ±SE	5.84^{a} ±0.11	5.69^{a} ±0.11	5.79^{a} ±0.11	5.78 ± 0.07
OC	upper	1.01^{b}± 0.48	2.05^{b}±0.48	3.89^{a}±0.48	2.32 ± 0.28
	middle	1.12^{b}±0.48	2.49^{ab}±0.48	3.04^{a}±0.48	2.22± 0.28
	lower	1.09^{b}±0.48	2.69^{ab}±0.48	3.07^{a}±0.48	2.29± 0.28
	Mean ±SE	1.07^{c} ±0.28	2.41^{b} ±0.28	3.33^{a} ±0.28	2.27 ± 0.16
TN	upper	0.09b±0.03	0.13b±0.03	0.21a±0.03	0.14 ± 0.02
	middle	0.08a±0.03	0.17a±0.03	0.18a±0.03	0.14± 0.02
	lower	0.08b±0.03	0.18b±0.03	0.21a±0.03	0.16± 0.02
	Mean ±SE	0.08^{b}±0.02	0.17^{a}±0.02	0.20^{a}±0.02	0.15 ± 0.01
C: N	upper	11.22b±2.02	15.89a±2.02	18.71a±2.02	15.27 ± 1.17
	middle	14.38a±2.02	13.83a±2.02	18.57a±2.02	15.59± 1.17
	lower	13.36a±2.02	14.59a±2.02	15.04a±2.02	14.44± 1.17
	Mean ±SE	12.98^{b} ±1.17	14.77^{ab}± 1.17	17.56^{a} ±1.17	15.11 ± 0.67
Av.P	upper	45.00a±4.73	18.63b±4.73	27.80ab±4.73	30.47 ± 2.73
	middle	27.83a±4.73	14.93a±4.73	13.10a±4.73	18.62± 2.73
	lower	54.17a±4.73	20.30b±4.73	23.03b±4.73	32.5± 2.73
	Mean ±SE	42.33^{a}±2.73	17.95^{b}±2.73	21.31^{b}±2.73	27.2 ± 1.58

Means in the rows followed by the same letter are not significant

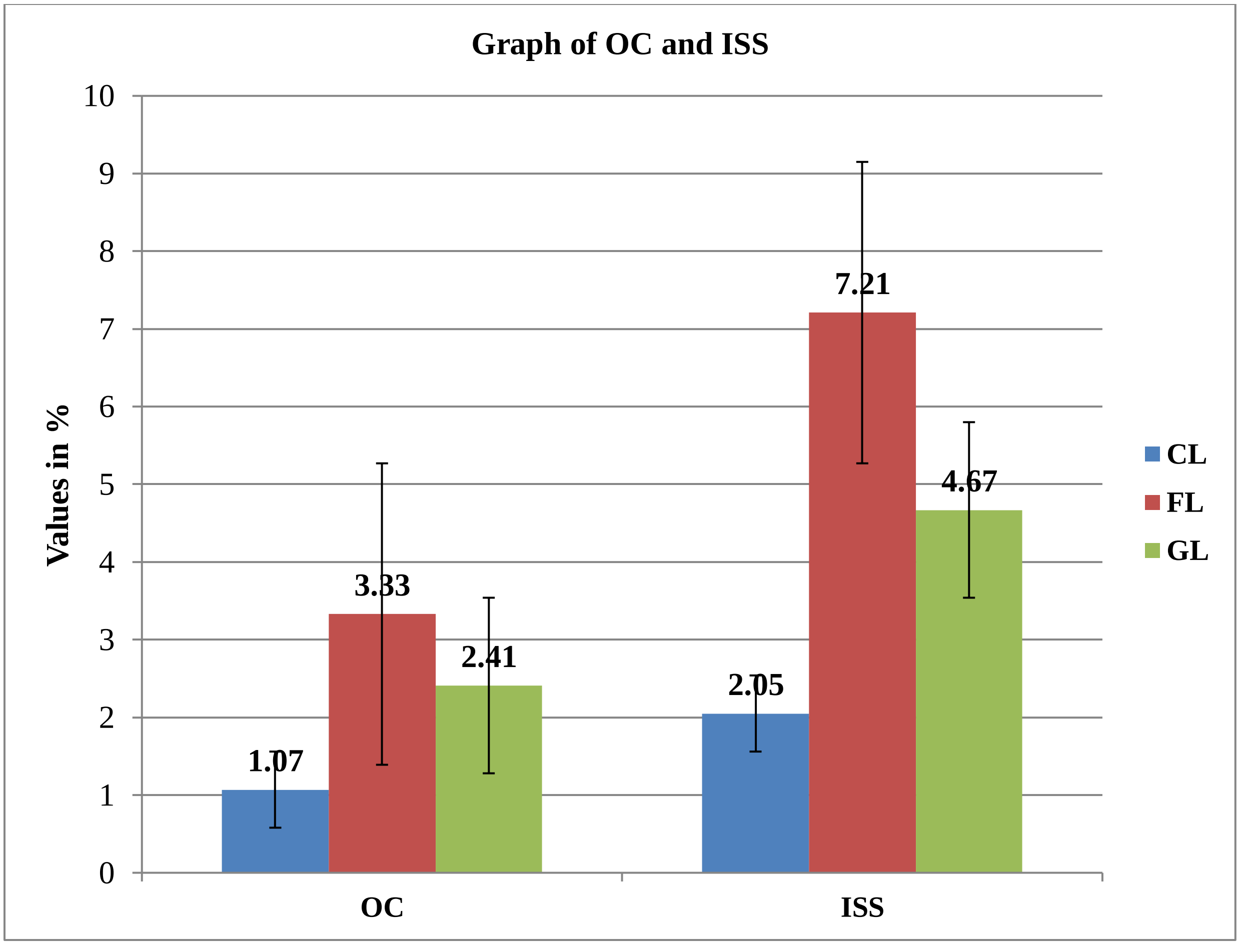
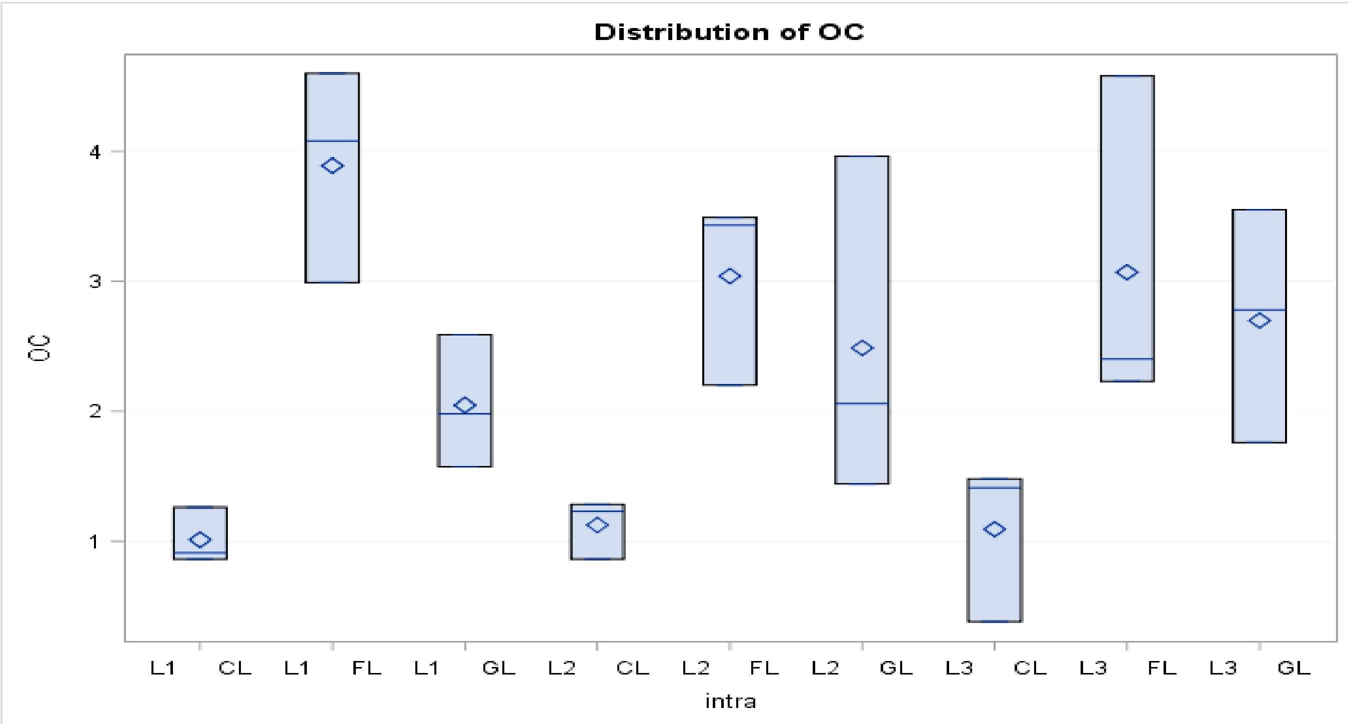


Fig 4.12. Graph showing the relationship between OC and ISS under different land use systems

where: OC, organic carbon; ISS, index of soil aggregate stability; CL, cultivated land; GL, grazing land and FL, forest land.

The highest value of available phosphorus was recorded in the cultivated land (42.33 mg kg^{-1}) followed by the forest land (21.31 mg kg^{-1}) whereas the lowest value was found in the grazing land (17.95 mg kg^{-1}). Significant differences ($p < 0.05$) were also observed in the mean values among locations; where the highest value was observed in the lower part of the watershed and the lowest value in the middle part where there is Vertisol coverage. Based on EthioSIS (2014) rating, the status of available phosphorus was low in the grazing and forest land use systems and moderate in the agricultural land; the overall mean value (27.2 mg kg^{-1}) was rated as low.



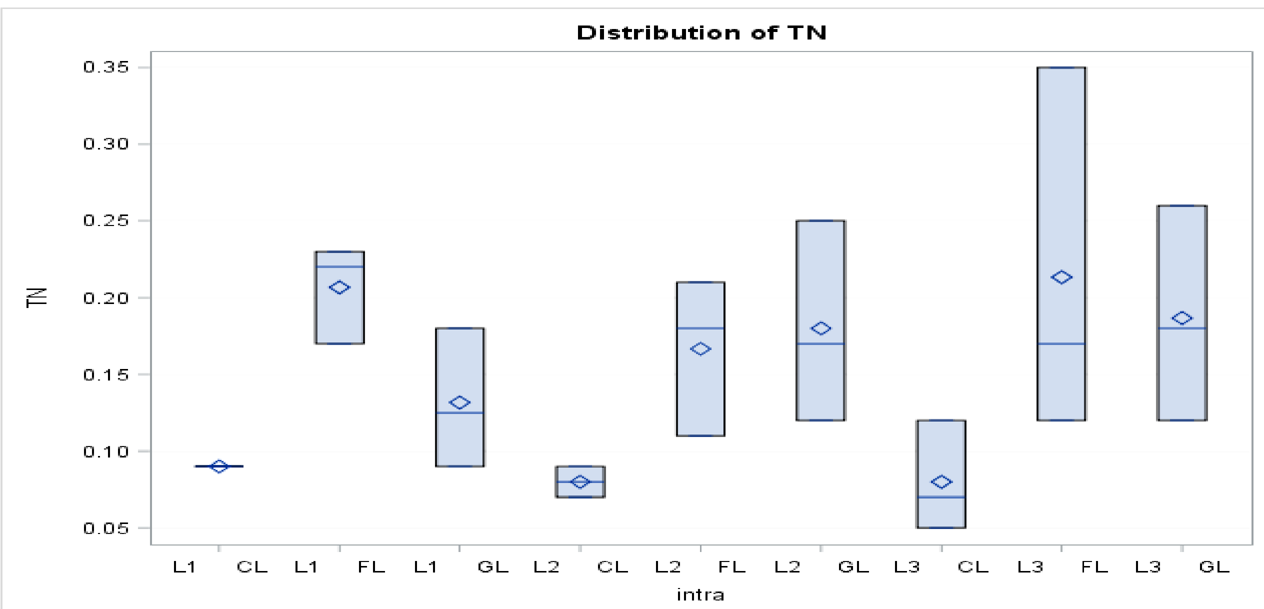
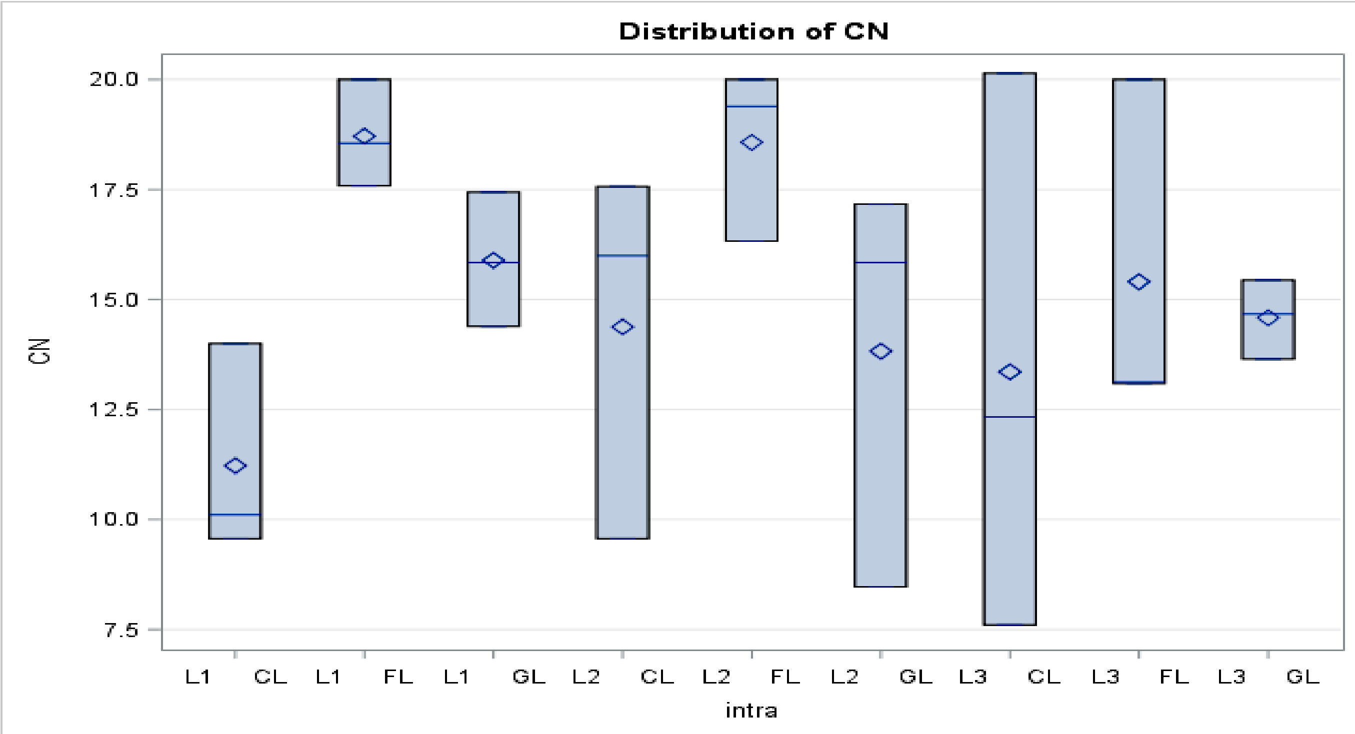


Fig 4.13. Interaction effects of location and land use (L*LU) on the distribution of (a) OC (%) and (b) TN

where, L1= location 1; L2 = location 2; L3 = location 3; CL = cultivated land; FL = forest land and GL = grazing land



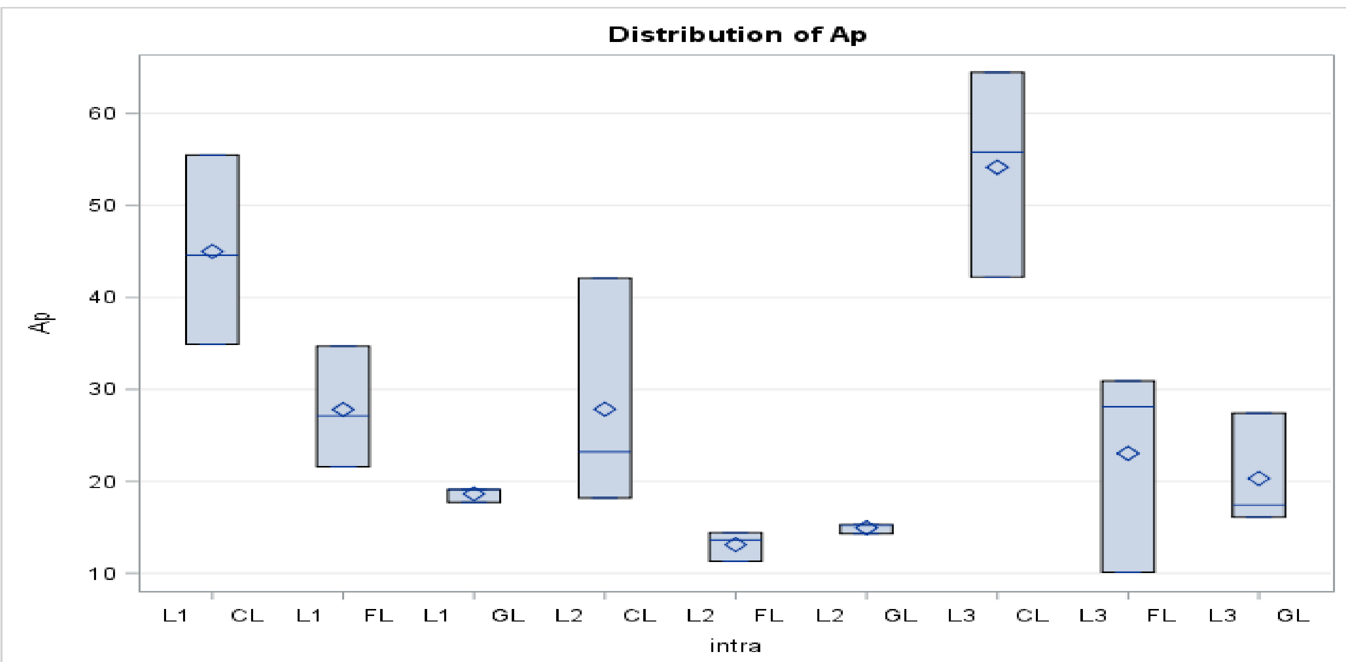


Fig 4.14. The Interaction effects of location and land use (L*LU) on the distribution of (a) C: N and (b) available P

where, L1= location 1; L2 = location 2; L3 = location 3; CL = cultivated land; FL = forest land and GL = grazing land

Exchangeable bases (Na^+, K^+, Ca^{2+} and Mg^{2+}), cation exchange capacity (CEC) and percent base saturation (PBS)

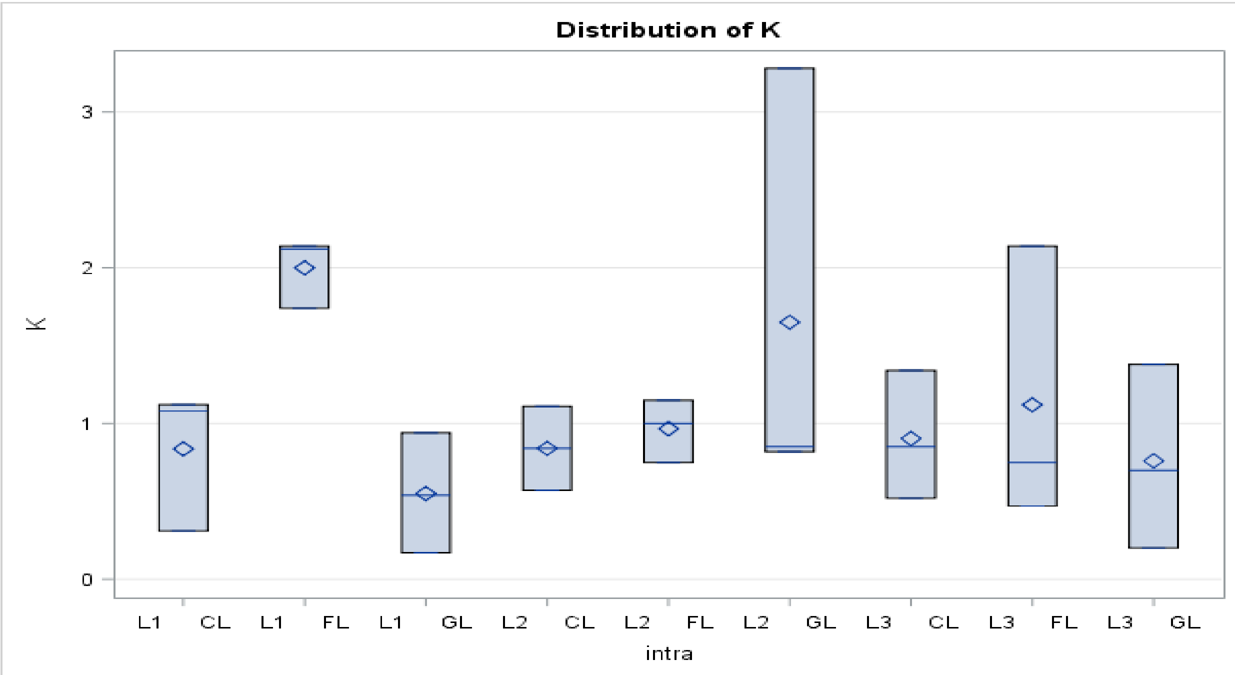
There is no significant difference in all exchangeable bases in the main and interaction effects of land use systems and elevation gradients (Table 4.18). However, marked variations were observed among land use systems. The lowest values of exchangeable Na^+ and K^+ were found in the cultivated land and the highest values in the forest land in all locations. The same is true for exchangeable Ca^{2+} and $mg^{2+.}$ Since the critical value of exchangeable Na^+ varied due to the variation in CEC, clay content and type of crop grown, its threshold value was more explained by ESP in which an ESP value of 15% is critical for most crops (Anon, 1954). The lowest (1.92%) and the highest (2.12%) values of ESP were observed in the grazing and forest land use systems respectively. Soils of the study area are free of sodicity problem as the values of ESP are far less than 15%. Based on the classifications of EthioSIS (2014), the contents of exchangeable bases are rated as high in all locations and land use systems.

The mean values of CEC didn't show significant differences between the main treatments (land use type and locations) and their interaction effects (LU*L) except in the mid land of the watershed where a significant difference was observed between cultivated land and forest land. Even though non-significant differences were found in the other cases, the numerical values were substantially varied between land use types and locations. The highest overall mean value (74.07 cmol (+) kg^{-1}) was found under the forest land followed by grazing land (68.37 cmol (+) kg^{-1}) and the lowest value (65.69 cmol (+) kg^{-1}) under the cultivated land. Base on (Landon, 2014) classifications, the contents of CEC were rated as very high in all land use systems and locations; indicating that soils are inherently fertile due to the parent materials from which they are developed (Elias, 2019; Williams, 2016). The highest mean value (86.12%) of PBS was observed in the forest land followed by grazing land (78.89%) and the lowest value was recorded in the cultivated field (74.13%). These values are rated as high and are indicators of good soil fertility.

Table 4.16. Mean values of exchangeable bases (Na^+, K^+, Ca^{2+} and Mg^{2+}), CEC and PBS under different land use systems and elevation gradient.

Soil property	**Elevation gradient**	**Land use system**			**Mean $\pm$SE**
		Cultivated land	Grazing land	Forest land	
Ex. Na	upper	$1.32^{a}\pm0.25$	$1.22^{a}\pm0.25$	$1.21^{a}\pm0.25$	1.25 ± 0.14
	middle	$1.51^{a}\pm0.25$	$1.20^{a}\pm0.25$	$1.52^{a}\pm0.25$	1.41 ± 0.14
	lower	$0.94^{a}\pm0.25$	$1.44^{a}\pm0.25$	$1.98^{a}\pm0.25$	1.46 ± 0.14
	Mean $\pm$SE	$1.25^{a}\pm0.14$	$1.29^{a}\pm0.14$	$1.57^{a}\pm0.14$	1.37 ± 0.08
Ex. K	upper	$0.84^{b}\pm0.38$	$0.55^{b}\pm0.38$	$2.00^{a}\pm0.38$	1.13 ± 0.22
	middle	$0.84^{a}\pm0.38$	$1.65^{a}\pm0.38$	$0.97^{a}\pm0.38$	1.15 ± 0.22
	lower	$0.90^{a}\pm0.38$	$0.76^{a}\pm0.38$	$1.12^{a}\pm0.38$	0.93 ± 0.22
	Mean $\pm$SE	$0.86^{a}\pm0.22$	$0.99^{a}\pm0.22$	$1.36^{a}\pm0.22$	1.07 ± 0.13
Ex. Ca	upper	$33.32^{b}\pm6.80$	$42.10^{a}\pm6.80$	$39.04^{a}\pm6.80$	38.15 ± 3.93
	middle	$28.82^{b}\pm6.80$	$33.45^{a}\pm6.80$	$39.90^{a}\pm6.80$	34.09 ± 3.93
	lower	$51.97^{a}\pm6.80$	$40.00^{a}\pm6.80$	$41.2^{a}\pm6.80$	40.85 ± 3.93
	Mean $\pm$SE	$34.48^{a}\pm3.93$	$38.53^{a}\pm3.93$	$40.07^{a}\pm3.93$	37.70 ± 2.27
Ex. Mg	upper	$13.28^{b}\pm2.53$	$15.93^{a}\pm2.53$	$11.05^{a}\pm2.53$	13.42 ± 1.46
	middle	$10.41^{b}\pm2.53$	$13.87^{a}\pm2.53$	$14.40^{a}\pm2.53$	12.89 ± 1.46
	lower	$21.42^{a}\pm2.53$	$13.96^{a}\pm2.53$	$13.21^{a}\pm2.53$	14.45 ± 1.46
	Mean $\pm$SE	$13.30^{a} \pm1.46$	$14.59^{a}\pm1.46$	$12.89^{a}\pm1.46$	13.59 ± 0.84
CEC	upper	$63.95^{a}\pm7.00$	$69.86^{a}\pm7.00$	$69.10^{a}\pm7.00$	67.63 ± 4.04
	middle	$59.62^{b}\pm7.00$	$66.17^{ab}\pm7.00$	$72.71^{a}\pm7.00$	66.17 ± 4.04
	lower	$74.04^{a}\pm7.00$	$69.08^{a}\pm7.00$	$80.43^{a}\pm7.00$	74.52 ± 4.04
	Mean $\pm$SE	$65.87^{a}\pm4.04$	$68.37^{a}\pm4.04$	$74.07^{a}\pm4.04$	69.44 ± 2.33
PBS	upper	$74.97^{a}\pm8.11$	$82.23^{a}\pm8.11$	$77.00^{a}\pm8.11$	78.06 ± 4.68
	middle	$70.23^{a}\pm8.11$	$75.17^{a}\pm8.11$	$78.43^{a}\pm8.11$	74.61 ± 4.68
	lower	$78.03^{a}\pm8.11$	$79.27^{a}\pm8.11$	$72.93^{a}\pm8.11$	76.76 ± 4.68
	Mean $\pm$SE	$74.13^{a}\pm4.68$	$78.89^{a}\pm4.68$	$86.12^{a}\pm4.68$	76.48 ± 2.70
	upper	2.22 ± 0.43	1.84 ± 0.43	1.73 ± 0.43	1.93 ± 0.25
ESP	middle	2.63 ± 0.43	1.86 ± 0.43	2.09 ± 0.43	2.19 ± 0.25
	lower	1.21 ± 0.43	2.11 ± 0.43	2.54 ± 0.43	1.95 ± 0.25
	Mean $\pm$SE	2.02 ± 0.25	1.94 ± 0.25	2.12 ± 0.25	2.03 ± 0.14

Means in the rows followed by the same letter are not significant



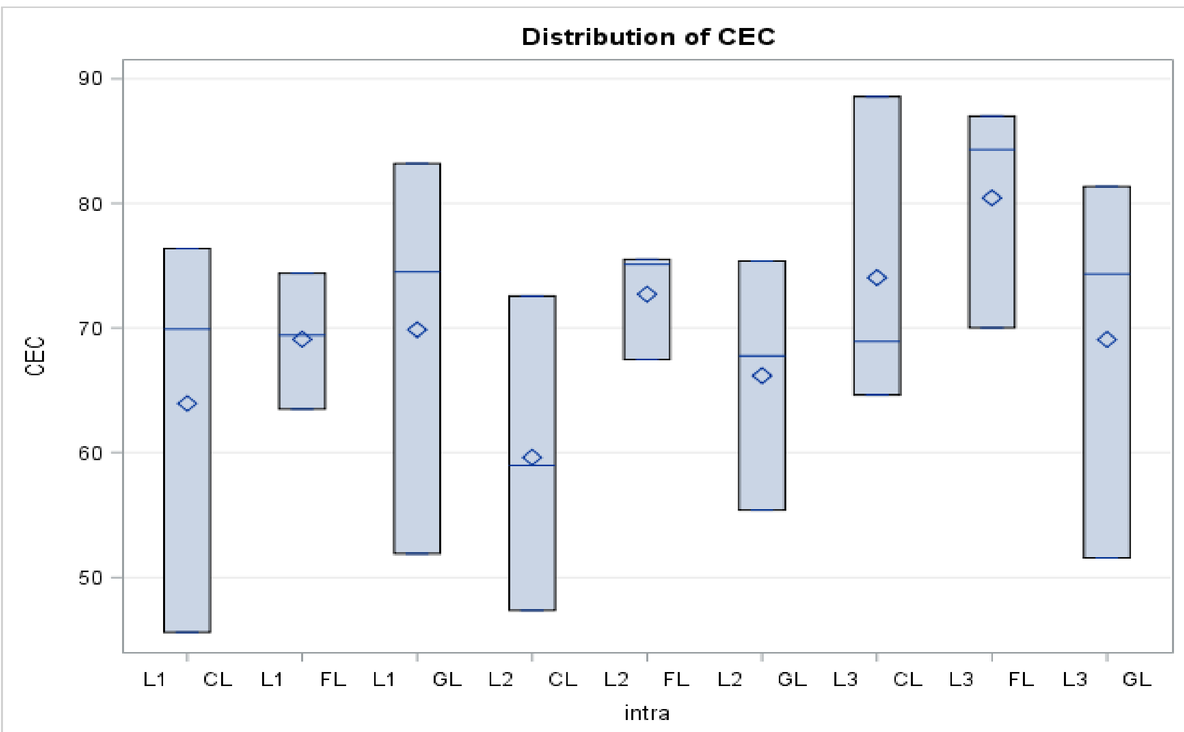


Fig 4.15. Interaction effects of location and land use type (L*LU) on the distribution of a} exchangeable K and b) CEC

Table 4.17. Post hoc multiple comparisons of statistically significant soil properties in contrasting land use systems

Soil quality indicator	Contrasted LU classes	P - value
ISS	Cultivated land and forest land	0.000
	Cultivated land and grazing land	0.000
	Forest land and cultivated land	0.000
	Forest land and grazing land	0.001
	Grazing land and cultivated land	0.001
	Grazing land and forest land	0.001
OC	Cultivated land and forest land	0.000
	Cultivated land and grazing land	0.003
	Forest land and cultivated land	0.000
	Forest land and grazing land	0.03
	Grazing land and cultivated land	0.003
	Grazing land and forest land	0.03
TN	Cultivated land and forest land	0.001
	Cultivated land and grazing land	0.008
	Forest land and cultivated land	0.001
	Grazing land and cultivated land	0.008
C: N	Cultivated land and forest land	0.013
	Forest land and cultivated land	0.013
Av.P	Cultivated land and forest land	0.000
	Cultivated land and grazing land	0.000
	Forest land and cultivated land	0.000
	grazing land and cultivated land	0.000

Table 4.17. Continued. Post hoc multiple comparisons of statistically significant soil properties in contrasted elevation gradients.

Soil quality indicator	Contrasted elevation gradients	P - value
FC	L2 and L3	0.05
	L3 and L2	0.05
PWP	L2 and L3	0.034
	L3 and L2	0.034
Av.P	L1 and L2	0.007
	L2 and L1	0.007
	L2 and L3	0.002
	L3 and L2	0.002

Table 4.18. Soil fertility ratings of some soil parameters used to identify the status of soil fertility (EthioSIS, 2014)

Soil properties	**Fertility classes (Ratings)**				
	Very low	low	Optimum	high	Very high
OM (%)	< 2.0	2.0-3.0	3.0-7.0	7.0-8.0	> 8.0
TN (%)	< 0.1	0.1-0.15	0.15-0.3	0.3-0.5	> 0.5
Av.P (mg kg^{-1})	0-15	15-30	30-80	80-150	> 150
Ex.K (cmol (+) kg^{-1})	< 0.23	0.23-0.49	0.49-1.54	1.54-2.31	> 2.31
	< 0.9	0.9-1.9	1.9-6.0	6.0-9.0	> 9.0
CEC cmol (+) kg^{-1})	< 4	4-15	15-25	25-40	> 40
Soil pH-H_2O	< 5.5 (Strongly acidic)	5.5-6.5 (Moderately acidic)	6.6-7.3 (Neutral)	7.3-8.4 (Moderately alkaline)	> 8.4 (Strongly alkaline)

Where: OM, organic matter; TN, total nitrogen; Av.P, available phosphorus and Ex.K, exchangeable potassium

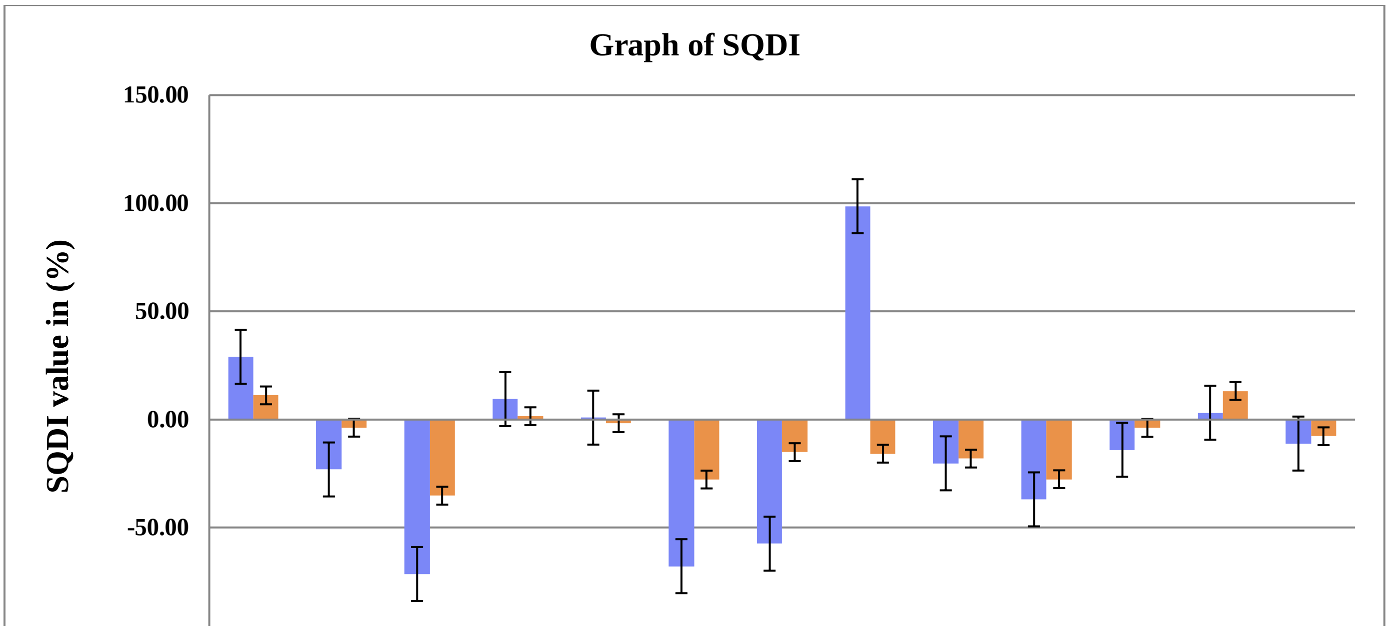
Table 4.19. Pearson's correlation matrix of selected soil physical and chemical properties

	Sand	Clay	Silt	FC	PWP	AWC	pH	OC	TN	C: N	Av.P	Na	K	Ca	Mg	CEC	PBS
Sand	1	-.755**	.455**	-.262	-.261	-.223	-.006	.029	-.075	.087	.118	-.182	.214	-.328*	-.364*	-.224	-.373*
Clay		1	-.705**	.441*	.932**	.470**	.210	-.096	-.066	.076	-.053	.173	-.147	.492**	.542**	.548**	.360*
Silt			1	-.577**	-.513**	-.614**	-.314	.455**	.459**	.022	-.330*	.191	.293	-.249	-.487**	-.453**	-.093
FC				1	.975**	.899**	.524**	-.023	-.003	.035	.308	.046	.056	.650**	.764**	.656**	.557**
PWP					1	.778**	.527**	-.005	-.039	-.045	.292	.003	.024	.650**	.748**	.611**	.509**
AWHC						1	.441*	-.056	-.085	.185	.294	.122	.111	.685**	.764**	.646**	.566**
pH							1	.116	.060	-.028	.338*	-.512**	.491**	.486**	.490**	.456**	.425*
OC								1	.908**	.487**	-.503**	.154	.645**	.459**	.060	.311	.399*
TN									1	.125	-.519**	.232	.625**	.419*	.069	.208	.451**
C: N										1	-.232	.128	.143	.337*	.169	.417*	.126
Av.P											1	-.294	-.186	-.057	.076	.073	-.183
Ex. Na												1	-.227	.082	-.011	.149	.011
Ex. K													1	.333*	.114	.132	.463**
Ex. Ca														1	.797**	.834**	.831**
Ex. Mg															1	.695**	.753**
CEC																1	.425*
PBS																	1

**, significant at $p < 0.01$ and *, significant at $p < 0.05$

4.3.3. Soil quality deterioration index (SQDI) for each quality indicator

Soil quality deterioration index results revealed that most of the soil quality indicators have negative values in the cultivated and grazing lands; indicating that the soil condition is becoming poor due to poor management in these land use systems. In the cultivated field, the lowest value was for ISS (-71.3%) followed by SOC (-67.7%). On the other hand, the highest positive value was for available P (98.6%) in the same land use system. For grazing land, the highest negative values were recorded for ISS (-35.1%) and SOC (-27.7%) (Fig 4.16). The aggregated SQDI values were -12.34% and -10.0 for cultivated land and grazing land respectively (Fig 4.17); these values were rated as low based on Wang et al. (2001) classification.



	Clay	Silt	ISS	AWC	pH	OC	TN	AP	Na+	K+	Ca2+	Mg2+	CEC
SQDI_CL	29.07	-23.05	-71.39	9.44	0.92	-67.77	-57.39	98.64	-20.21	-36.87	-13.96	3.20	-11.08
SQDI_GL	11.19	-3.73	-35.17	1.56	-1.67	-27.70	-15.06	-15.75	-18.02	-27.57	-3.85	13.24	-7.70

Fig 4.16. SQDI values of soil quality indicators under cultivated and grazing lands using forest land as a reference.

Note: SQDI is soil quality deterioration index, CL is cultivated land and GL is grazing land

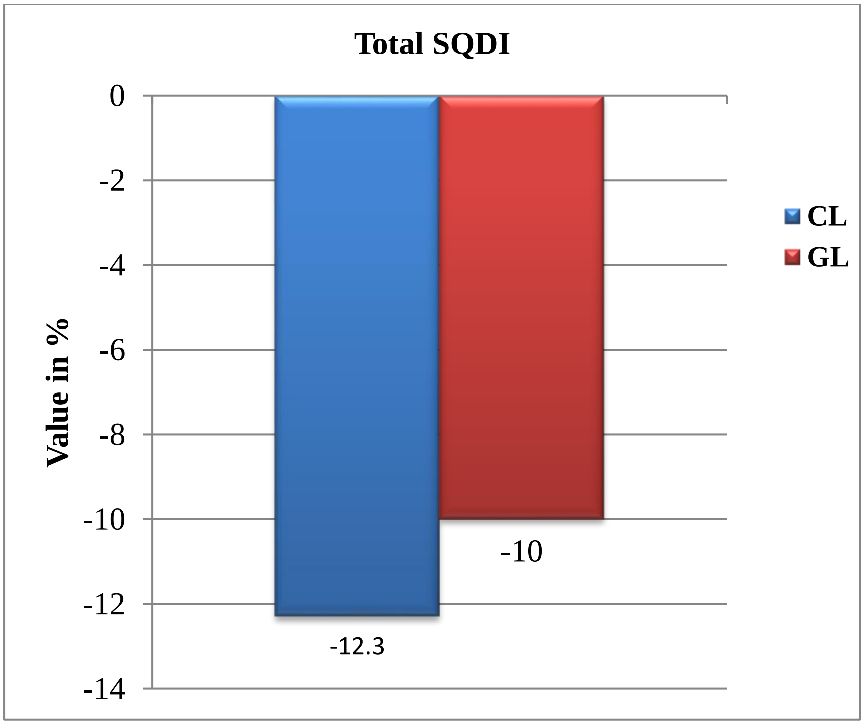


Fig 4.17. Aggregated SQDI value for cultivate land (CL) and grazing land (GL) in Suha watershed

Soil quality index (SQI) computation from production and soil erosion perspectives

The overall SQI mean value under the cultivated land use system was significantly lower than the forest and grazing lands. The mean values were 0.38, 0.60 and 0.52 for cultivated land, forest, land and grazing land respectively (Fig 4.18). Based on Li et al. (2018) classification, the soil quality was rated as low, moderate and high for cultivated land, grazing land and forest land respectively (Table 4.22).

Table 4.20. Status of soil quality under different land use systems based on Li et al. (2018) ratings

Land use	SQI values for selected indicators				Total SQI	Rating
system	STC	pH	OC	NPK		
CL	0.04	0.06	0.16	0.12	0.38	Low
FL	0.04	0.06	0.32	0.18	0.60	High
GL	0.04	0.06	0.24	0.18	0.52	Moderate

Note: CL = Cultivated land, FL = Forest land, GL = Grazing land, SQI = Soil quality index, STC = Soil texture, OC = Organic carbon and NPK = Nitrogen, phosphorus and potassium.

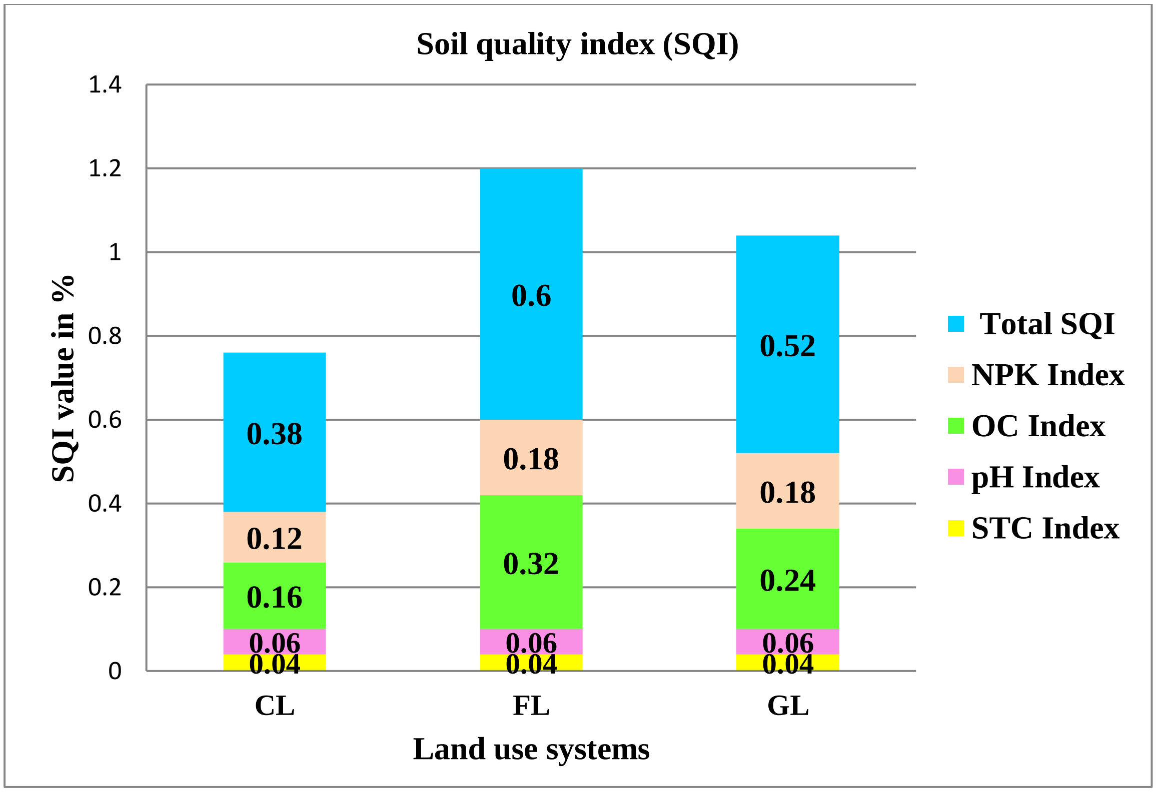


Fig 4.18. Soil quality index (SQI) values under different land use systems from production and soil erosion perspectives

4.4. Quantify soil nutrient flows and balances (N and P) in cereal-based agroecosystems of Suha watershed, northwestern Ethiopia

4.4.1. Nutrient inflows and outflows in different agroecosystems

The results of soil nutrient inputs to the system through the applications of mineral fertilizer, organic input, atmospheric deposition and N-fixation and nutrient exports through crop harvest (OUT1 and OUT2), nutrient loss by soil erosion, leaching, and gaseous losses of the two farming systems are presented in (Table 4.25 and 4.26). The main sources of nutrient inputs for N and P were mineral fertilizers which include NPSB (18.9% N, 37.7% P2O5, 6.95% S, and 0.1% B) and Urea (46% N). The amount of N used ranged from 87.9 kg N/ha/yr to 129.5kg N/ha/yr in the highland and from 64.9 kg N/ha/yr to 97.4 kg N/ha/yr in the midland. The amount of P ranged from 17 kg P/ha/yr to 33 kg P/ha and from 17 kg P/ha/yr to 25 kg P/ha for highland and midland respectively. The utilization of mineral fertilizers for cereal crops has been well adopted in the area regardless of its ever-increasing price; which is unaffordable for resource-poor farmers. However, the utilization of only mineral fertilizers does not guarantee the sustainability of soil fertility. Potassium (K) containing fertilizers were not applied in the study area assuming that soils of the highland regions contain as sufficient amount of K (Elias et al., 1998). Organic inputs (IN2) were not applied to the farmlands which are far from the residential area because of their competitive use and labor demand for transportation.

Atmospheric deposition (IN3) and non-symbiotic N-fixation were the other nutrient inflows accounted for in this study. Estimated values of atmospheric deposition for the highland area were 5.2 kg/ha/yr; 0.8 kg/ha/yr and 3.4k g/ha/yr for N, P and K respectively. In the midland area, the values were 4.7 kg/ha/yr, 0.8 kg/ha/yr and 3.1 kg/ha/yr for the same nutrients. The amount of N fixed by non-symbiotic bacteria was 4.2 kg N/ha/yr in the highland area and 3.9 kg N/ha/yr in the midland area of the watershed. Moreover, the results of nutrient inputs (IN1-IN4) showed significant differences among socioeconomic groups and farming systems.

The outflow results revealed that, crop harvest (OUT1 and OUT2) was one of the most important pathways for the removal of N, P and K from all agroecosystems. The contents of N and K in crop harvest (yield and residues) were higher; whereas, the export of P from the system through this process was far lower than N. The largest proportion of N was exported with yield; whereas, removal with residue was the major factor for K. The mean values of N and P exported with yield and residue were 75.8 kg N/ha/yr and 28.9 kg N/ha/yr; 14.6 kg P/ha/yr and 12.1kg P/ha/yr respectively for the wheat farm land. For teff farming system, the values of N and P in the crop yield were 42.1 kg N/ha/yr and 8.3 kg P/ha/yr respectively. The values in the residue were 29.1 kg N/ha/yr and 8.7 kg P/ha/yr.

Estimated results of nutrient losses through leaching and denitrification processes showed that a significant amount of N was lost from the farming systems. It was observed that 30.5 kg/ha/yr of N was lost from the wheat field; whereas, 25.2 kg/ha/yr of N was lost from the teff field through the leaching process. The amount of exported nutrients through leaching depends on the soil type, climatic condition and the amount and type of soluble nutrients found in the soil (Haileslassie et al., 2005). The values of N loss through denitrification were 16 kg N/ha/yr and 11.9 kg N/ha/yr for the wheat and teff fields respectively. Results also showed that denitrification is the least contributor to N loss from the farming system.

Soil erosion was one of the most important factors in nutrient removal from the system. The results showed that the values of exported nutrients were higher in the highland area as compared to the midland part of the watershed. Quantified values indicated that 48kg N /ha/yr and 10.6 kg P /ha/yr were removed by soil erosion. In addition, significant differences were observed in nutrient outputs (OUT1-OUT5) among socioeconomic groups and agroecosystems.

4.4.2. Nutrient balances

The results of nutrient balances (N and P) of different farming systems (highland and midland areas) were depicted in (Tables 4.27; Fig 4.21 and 4.22). The balance of N was negative in all farming systems regardless of socioeconomic groups. The mean values of N and P balances were -77.1 kg N/ha/yr and -11.9 kg P/ha/yr respectively in the highland area (wheat farming system); whereas in the midland (teff farming system), the values were -39.3 kg N/ha/yr and -1.4 kg P/ha/yr. Significant variations were also observed among socioeconomic groups and agroecosystems. It was observed that the balances of N were more negative in all farming systems and both agroecologies. Moreover, the values were more negative in the highland areas than the midland areas. Significant differences were observed in the balance of N among socioeconomic groups and between agroecologies. On the other hand, P balances were less negative in all farming systems and agroecologies regardless of the socioeconomic groups (Fig 4.21 and 4.22). Even though the values were less negative, considerable differences were observed among socioeconomic groups and agroecologies. The analysis of variance results also showed that significant differences were observed in the values of P balances among socioeconomic groups and agroecologies.

Table 4.21. Nutrient inputs in different farming systems and socioeconomic groups

Agroecology	Wealth group	IN1		IN2		IN3		IN4	Total (IN1 + IN2 + IN3 + IN4)	
		N	P	N	P	N	P	N	N	P
Highland	R	129.5	33	0	0	5.2	0.8	4.2	139	33.3
	M	120.9	25	0	0	5.2	0.8	4.2	130	25.4
	P	87.9	17	0	0	5.2	0.8	4.2	97	7.4
Midland	R	97.35	25	0	0	4.7	0.8	3.9	106	25.6
	M	87.9	21	0	0	4.7	0.8	3.9	97	21.5
	P	64.9	17	0	0	4.7	0.8	3.9	74	17.4

Where, IN1= input 1; IN2= input 2; IN3 = input 3 IN4 = input4; R = rich; M = medium and P = poor

Table 4.22. Nutrient outputs in different farming systems and socioeconomic groups

Outflows	Highland (wheat farming system)			Midland (teff farming system)		
	R	M	P	R	M	P
OUT1						
- N	95.9	78.1	53.5	51.9	41.5	32.9
- P	18.5	15.1	10.3	10.2	8.2	6.5
OUT2						
- N	36.1	29.9	20.6	34.2	30.3	22.9
- P	15.1	12.5	8.6	10.1	9.1	7
OUT3						
- N	32.9	32.7	26.2	27.8	26.6	21.1
OUT4						
- N	18.2	17.1	12.8	13.8	12.6	9.6
OUT5						
- N	48	48	48	23	23	23
- P	10.6	10.6	10.6	6.2	6.2	6.2
Total (OUT1+ OUT2 + OUT3 + OUT4 + OUT5)						
- N	231.1	205.7	161.1	150.6	133.9	109.5
- P	44.2	38.1	29.5	26.4	23.4	19.7
Balances						
- N	-92.1	-75.4	-63.8	-44.7	-37.4	-35.9
- P	-10.8	-12.7	-12.1	-0.8	-1.9	-2.4

Table 4.23. Aggregated nutrient flows and balances of wheat farming system in the highland of the watershed

Nutrient flows	N	P	Remark
IN1	112.8	25	
IN2	0	0	
IN3	5.2	0.8	
IN4	4.2	0	
Sum (IN1+IN2+IN3+IN4)	122	25.4	
OUT1	75.8	14.6	
OUT2	28.9	12.1	
OUT3	30.5	0	
OUT4	16	0	
OUT5	48	10.6	
Sum (OUT1+OUT2+OUT3+OUT4+OUT5)	199.2	37.3	
Partial nutrient balance	+8.1	-1.7	(IN1+ IN2)-(OUT1 + OUT2)
Full nutrient balance	-77.1	-11.9	(IN1+IN2+IN3+IN4)-(OUT1 + OUT2 + OUT3+OUT4+OUT5)

Where, OUT1= out put1;OUT2 = out put2;OUT3 = out put3;OUT4 = out put4 and OUT5 = out put5

Table 4.23. continued. Aggregated nutrient flows and balances of the Teff farming system in the midland of
the watershed

Nutrient flows	N	P	Remark
IN1	83.4	21	
IN2	0	0	
IN3	4.7	0.8	
IN4	3.9	0	
Sum (IN1+IN2+IN3+IN4)	92	21.8	
OUT1	42.1	8.3	
OUT2	29.1	8.7	
OUT3	25.2	0	
OUT4	11.9	0	
OUT5	23	6.2	
Sum (OUT1+OUT2+OUT3+OUT4+OUT5)	131.3	23.2	
Partial balance	+12.2	+4	(IN1+IN2)-(OUT1+OUT2)
Full nutrient balance	-39.3	-1.4	(IN1+IN2+IN3+IN4)- (OUT1 + OUT2 +OUT3+OUT4+OUT5)

4.3.1. Replacement cost of depleted nutrients

From the full nutrient balance results, it is observed that a substantial amount of N and P were removed from the wheat and teff cultivated fields. The amounts of nutrients depleted were converted in to fertilizer and multiplied by the current price to calculate replacement costs. Monetary values were 1807 and 3400 ETB/ha/yr for NPSB and Urea fertilizers respectively for the high land 212.6 ETB/ha/yr and 1899.8 ETB/ha/yr mid land respectively.

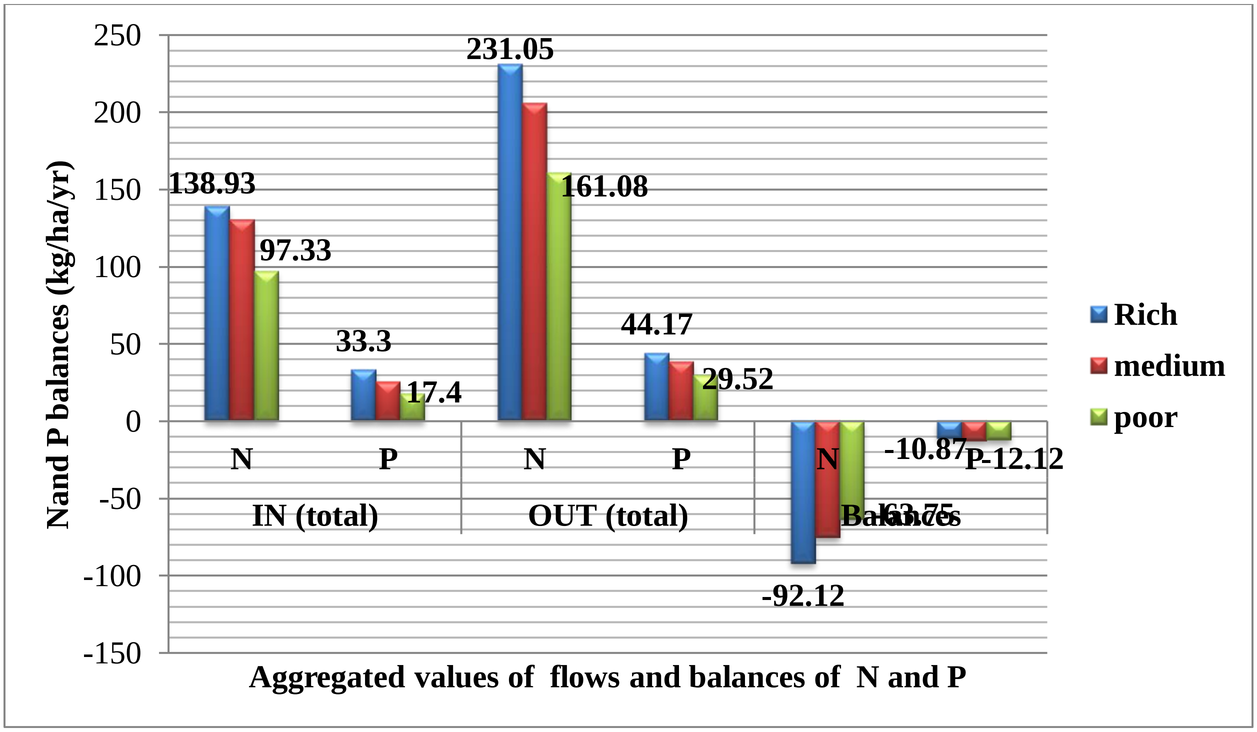


Fig 4.21. Aggregated values of nutrient flows and balances in the highland of the watershed

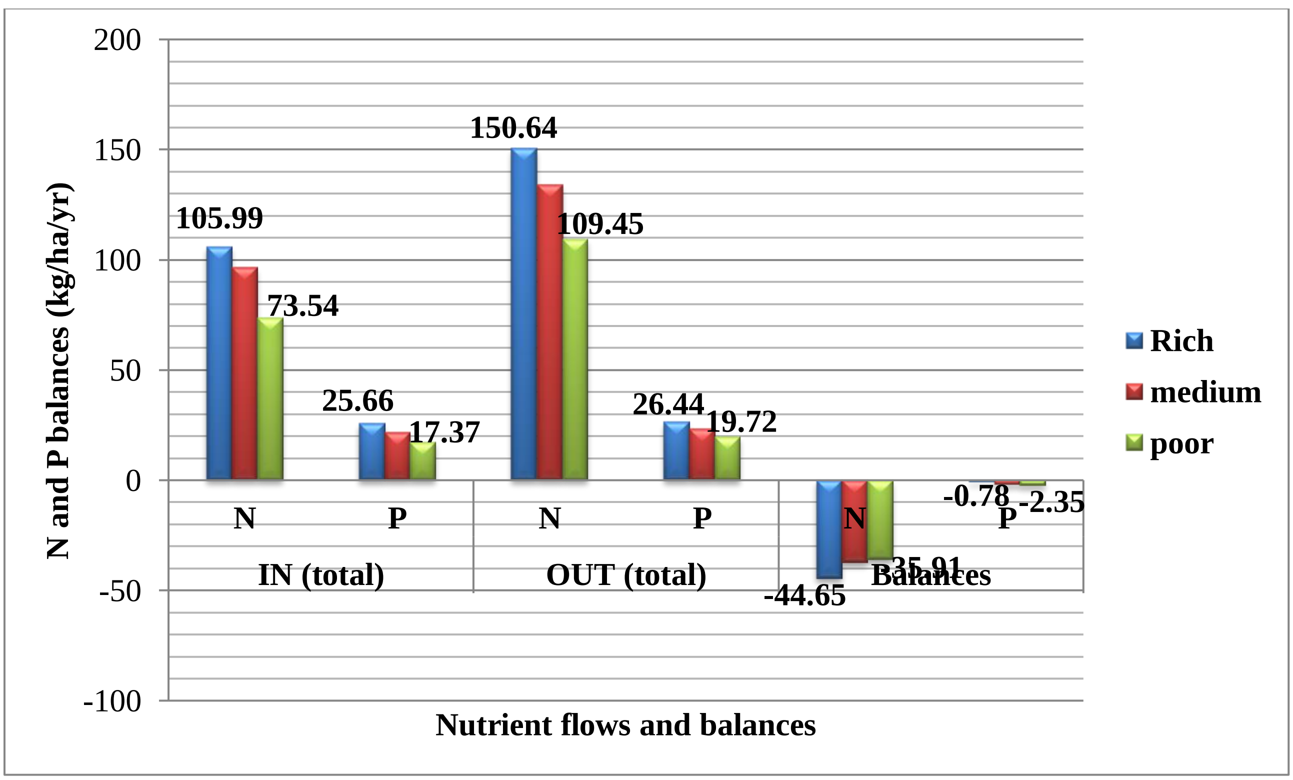


Fig 4.22. Aggregated values of nutrient flows and balances for N and P in the midland of the watershed

CHAPTER FIVE: DISCUSSION

5.1. Detection of LULC and LST changes using GIS and remote sensing in Suha Watershed

Land use/cover changes in Suha watershed

Anthropogenic activities continuously alter the land surface features. The results of this study showed that significant land use land cover changes occurred in the study area for the last 35 years. Agricultural lands increased as the expense of other land use types due to proximity and root causes. There is high demand for agricultural products and fuel wood because of the ever-increasing population in the area. This creates greater pressure on the natural resource base. Grass land decreased during this time, because part of it was distributed to landless youths. Shrub lands are continuously and illegally cleared by the local community for fire wood. These uncontrolled humane interferences further aggravate the degradation of environmental resources. Similar findings were reported by (Hassen and Assen, 2018). Aneseyee et al. (2020) also reported agricultural land, built-up area and bare land were expanded by 33.01%, 109.58% and 65.19% within a 30 years period in Winka watershed, Ethiopia. On the contrary, grazing land declined by 49.12% in the same period. Belenok et al. (2021) analyzed landscape changes (1985-2020) and explored that the built-up area expanded significantly.

The reasons for the significant reduction of shrub land and grazing lands in the study periods might be due to land reform (redistribution of farm lands) and continuous distribution of communal grass lands to youths and landless farmers. Due to this action, even farmlands became more fragmented. The state change that took place in Ethiopia (the fall of the 'Derge' regime and the rise of the 'EPRDF') brought fundamental changes to land ownership in the study area. The results also showed an increasing trend of forest land from 1999 to 2019. This could be attributed to an expansion of eucalyptus plantation (*Eucalyptus globules*) at the homestead, roadside plantation, farm boundaries, and even woodlot plantation in their farmlands (when agricultural lands are becoming exhausted because of poor soil conditions). The root cause for this expansion is an increasing human population that creates greater demand for basic needs (constriction materials and fuel wood). This result is consistent with previous finds (Gedefaw et al., 2020; Alemu et al., 2015**;** Hassen and Assen, 2018; Teferi et al., 2013). Such types of local-level studies are useful in understanding the status of other environmental issues, such as soil erosion, landslide risk, land use planning, carbon sequestration, land surface temperature, and ecosystem service which are essential for decision-making and planning purposes (Damtea et al., 2020).

Impacts of land use/cover changes on land surface temperature in the Suha watershed

Changes in the biophysical compositions of the earth's surface brought a change in LST (Rajani and Varadarajan, 2021). The same author also confirmed that land use land cover changes significantly affect land surface temperature in space and time. Analyzed results of LST and LULC changes showed that impervious surfaces have the highest LST values whereas forest and shrub lands have the lowest LST values. This variation could be attributed to the characteristics of land use and land cover features. These results are in line with the findings of Haylemariyam (2018); Imran et al. (2021).

Areas covered with forests have low LST value due to the cooling and balancing effects of forests (Alemu, 2019; Haylemariyam, 2018). Gémes et al. (2016) also reported that in highly vegetated areas, the surface temperature can be reduced by 13^{o}C as compared to none vegetated areas and thus play a great role in human health conditions. On the other hand, the characteristics of impervious surfaces are the major factor for land surface temperature increment. LST is significantly increasing in impervious surfaces such as built-up areas and bare lands because these surfaces absorb radiant energy and stored heat instead of reflection or transmission. In other words, heat is retained for a longer time in these land features compared with other LULC classes (Shastri et al., 2020). Furthermore, land use land cover change considerably impact air humidity which is related to LST; impervious surfaces lack humidity and vegetation area has a high value (Igun and Williams, 2018; Awuh et al., 2019). Alemu (2019) also demonstrated that impervious layers are experiencing higher land surface temperature. Results of this study also confirmed that, areas which were previously categorized under lower LST values were converted in to higher LST classes due to the Expansion of bare lands and built-up areas during the study periods. From this, it can be inferred that LU/LC classes significantly impact LST compared with other factors that can affect LST. Therefore, continuous LU/LC changes particularly changes from forest area to impervious surfaces increase the land surface temperature. Impacts of LST can be manifested by changes in the environment, ecosystem service and human thermal comfort in urban areas, biodiversity, hydrology, and climate (Awuh et al., 2019; Das and Angadi, 2020; Imran et al., 2021).

Correlation between NDVI and LST

The results of this study revealed that normalized difference vegetation index (NDVI) has a negative correlation with LST. In this regard, the highest NDVI and the lowest LST values were found in forest areas and lowest NDVI and higher LST were found under impervious layers. In the study periods NDVI and LST have a negative correlation with R^2 values ranging from - 0.865 to - 0.914. The correlation coefficients of NDVI and LST in this research represent a very strong negative correlation between these two parameters based on the strength of the correlation coefficient suggested by Cohen, West and Aiken (2013). Accordingly, values between 0.31 and 0.5 describe weak correlation, from 0.51 to 0.7 describe normal correlation, from 0.71 to 0.90 describe strong correlation and from 0.91 to 1.0 describe very strong correlation. Similar findings have been reported by Traore et al. (2021); Hua and Ping (2018). Results depicted in Fig. 6 and 8 also clearly showed that vegetation has stronger negative correlation than other LU/LC classes because of its higher NDVI value. Similar finding were also reported by (Haylemariyam, 2018; Imran et al., 2021).

5.2. Quantification of soil erosion and sediment yield in response to land use and land cover changes using GIS and RUSLE model

Soil erosion rate and spatial distribution in Suha watershed

Soil erosion causes critical socio-economic and environmental problems, particularly in developing countries. To design and implement successful management strategies, it is necessary to understand the mechanisms, extent, and severity of soil erosion, as well as the causes that drive it. Monitoring the spatiotemporal variability of soil erosion risk and sediment export is required for two fundamental practical reasons. The first one is, for proper resource allocation and effective watershed management activities. Soil and water conservation activities are labor intensive and demand huge investment. Therefore, available limited resources should be allocated to those landscapes which are vulnerable to soil erosion. The second reason is, for selection of effective soil and water conservation technologies. Different soil and water conservation technologies could be recommended for specific location depending on the agro ecology, topography and land use systems.

The results of this study revealed that the magnitude of mean annual soil loss rate significantly increased from 15.2 t ha^{-1} yr^{-1} to 30.2 t ha^{-1} yr^{-1} over the past 35 years (1985 - 2019). Besides to this, the annual gross soil loss from the entire catchment similarly increased from 1.222 million tons per year to 2.426 million tons per year. This could be attributed to an expansion of cultivated land and bare land as the expense of other land use types during the study period. The presence of steep slope landscapes, poor land management systems and absence of soil and water conservation strategies are also contributing factors for soil loss. Moreover, the variation of the amount of rainfall is one major cause for higher soil loss and sediment yield in the watershed. Soil formation rate and soil loss tolerant limit for Ethiopian condition were suggested by Hurni (1986), which is within the range of 2-22 t ha^{-1} yr^{-1} and 2-18 t ha^{-1} yr^{-1} respectively. However, the values of this study are greater than these limits. In addition, the mean annual soil loss rate greater than 10 t ha^{-1} yr^{-1} could not be reversed in 50 years (Kouli et al., 2009). The results of this study are greater than this limit which confirmed the risk of soil erosion in the Suha watershed. The trend and spatial distribution of soil erosion risk is highly impacted by human-induced activities. The highest amount of soil loss and sediment yield was observed from steep slope areas (dominated by bare lands), cultivated fields on sloppy areas and open shrub lands. On the other hand, the lowest values were found in gentle slope areas and forest land use systems. Soil erosion risk is extremely high in 2009 as compared to other study periods due to the variation in mean annual rain fall and its erosive power. Average annual rainfall has the highest value (1443.7mm) in 2009 compared to other periods; in 1985, the average annual rainfall is 1186mm; in 1999 (1324.3mm), and in 2019 (1317.6 mm). Haregeweyn et al. (2015) conducted a review work on soil erosion and conservation in Ethiopia and inferred that 35% of the spatial and temporal variability of soil erosion is because of the variation in rainfall. Besides this, land use/cover dynamic is a major factor for the change in soil loss rate and sediment yield. This continuous severe soil loss and sediment yield causes both onsite and off-site effects of soil erosion. Transport of soil nutrient elements with sediment which intern impacts agricultural productivity and food security is an onsite effect of soil erosion. In addition, sediment deposition and eutrophication on water bodies, particularly on lakes is an off-site effect.

The results of this research are within the range of the findings of previous studies conducted in the Upper Blue Nile Basin and other parts of Ethiopia (Table 4.13). The results of the present study are comparable with the findings of Fenta et al. (2021), who reported a mean annual soil loss rate of 32.8 t/ha/yr from the Abay Basin. Similarly, Degife et al. (2021) reported a soil loss rate of 37 t/ha/yr from the Hawassa Lake catchment, Ethiopia. Haregeweyn et al. (2017) from the

Upper Blue Nile River Basin (27.5 t/ha/yr); Kinde et al. (2019) from Guder sub watershed, central highlands of Ethiopia (25-30 t/ha/yr); Atoma et al. (2020) from Huluka watershed, central Ethiopia (14.4-27 t/ha/yr); and Tadesse and Abebe (2014) from Jabi Tehinan watershed, northern Ethiopia (30.4 t/ha/yr) reported similar results. On the contrary, the findings of some other studies from different parts of Ethiopia showed a higher soil loss rate than the current study. For example, Tamene et al. (2017) reported a mean annual soil loss rate of 48t/ha/yr from the Laygeda watershed, Ethiopia. The findings of Zerihun et al. (2018) from Dembecha district showed soil loss tare of 49t/ha/yr. The study of Gelagay and Minale (2016) indicated soil loss rate of 47t/ha/yr from the Koga watershed; Belayneh et al. (2019) reported a soil loss rate of 42.8t/ha/yr from the Gumara watershed. The recent study by Woldemariam and Harka (2020) showed the soil loss rate of 75.85 t ha^{-1} yr^{-1} (in 2000) and 107.07t/ha/yr (in 2018) from Erer Sub-Basin, Wabi Shebelle Basin, Ethiopia. In the same study, it was reported that soil erosion severity class increased by 18.28% over the past 18 years. On the other hand, some other studies reported small values of soil loss rate compared to this study. For instance, Bekele and Gemi (2021) reported a mean annual soil loss rate of 2.2t/ha/yr from the Dijo watershed, Rift Valley Basin. Similarly, the results of Tiruneh and Ayalew (2015) from the Enfranz watershed (4.8 t/ha/yr); Ayalew (2015) from the Zingin watershed (9.1 t/ha/yr) and Brhane and Mekonnen (2009) from Medego Watershed (9.6 t/ha/yr) are smaller than the results of the current study. The variation of the mean annual soil loss rate in different watersheds could be due to the variation in topography, management (land use system), and the amount of rainfall.

Sediment delivery ratio (SDR) and Sediment export

SDR was estimated using channel bed slopes and the results showed that SDR has direct proportion with channel bed slope. As the channel bed slope increases, the velocity of runoff and sediment export also increase from the catchment. From the results of the three study periods, it is observed that the highest value of sediment yield (8.16 t ha^{-1} yr^{-1}) was recorded in 2009 compared to the other three periods; 3.95 t ha^{-1} yr^{-1} in (1985); 5.66 t ha^{-1} yr^{-1} (in 1999) and 8.02 t ha^{-1} yr^{-1} (in 2019). This could be due to detrimental impacts of LULC changes in which cultivated land and bare land increased as the expense of other land use types. The results of the current study are within the ranges of previous findings. For instance, Fenta et al. (2021) reported 7 t ha^{-1} yr^{-1} mean sediment yield from Abay Basin. Kidane et al. (2019) estimated sediment yield for the three periods (1973, 1995 and 2015) in the Guder sub watershed, Ethiopia and their results showed that the mean sediment yields were 6.79, 8.65 and 9.44 t ha^{-1} yr^{-1}. Tamene et al. (2017) from Laygeda watershed, Ethiopia (12.3 t ha^{-1} yr^{-1}) and Haregeweyn et al. (2017) from the upper Blue Nile Basin (7.34 t ha^{-1} yr^{-1}) reported similar findings.

Table 5.1. Some of the recent research findings of soil loss and sediment yield in different catchments of Ethiopia.

No.	Catchment/ study area	Method employed	Men annual soil loss (t/ha/yr)	Sediment yield (t/ha/yr)	Reference
1	Abay Basin	RUSLE	32.8	7	Fenta et al.(2021)
2	Hawassa lake catchment	InVEST	37	1.6	Degife et al. (2021)
3	Huluka watershed	RUSLE	27	-	Atoma et al. (2020)
4	Guder sub watershed	RUSLE	30.25	9.44	Kidane et al. (2019)
5	Gumara	RUSLE	42.67	-	Belayneh et al. (2019)
6	Beshillo catchment	RUSLE	37	-	Yesuph and Dagnew (2019)
7	Upper Blue Nile Basin	RUSLE	27.5	7.34	Haregeweyn et al. (2017)
8	Laygeda watershed	RUSLE	48	12.3	Tamene et al. (2017)

Soil erosion severity classes based on sub-watersheds

Exploring the spatial variability and identifying soil erosion risk areas is crucially important for technology selection; proper planning, resource allocation and application of soil and water conservation strategies on the bases of severity class. In this study, soil erosion severity classes were identified by superimposing the soil loss raster maps and sub-watershed maps. Accordingly, sub-watersheds were categorized into five soil erosion severity classes ranging from very slight to very severe on the basis of the rates of mean annual soil loss. SW15, SW 17 and SW18 were classified under the very severe soil erosion severity class (soil loss rate > 50 t/ha/yr) and found on the lower part of the watershed. These watersheds cover 7611.5ha of the total area of the watershed and are present on steep slope areas. They are the priority areas that demand urgent soil and water conservation measures. Sub-watersheds under severe soil erosion severity class (soil loss rate ranged from 30-50t/ha/yr) include, SW1, SW2,SW3, SW4, SW13, SW14, SW16, SW20, SW21, SW22 and SW23 and cover 35,375.2ha from the total watershed area. These sub-watersheds are the second priority areas to undertake management actions. The other sub-watersheds which cover 37,3563 of the watershed area are the last priorities for soil conservation activities as evidenced by the results of soil loss rates from these areas (Table 4.14).

Table 5.2. Soil erosion severity class and priority levels of sub-watersheds

Soil loss rate (t/ha/yr)	Severity class	Priority level	Sub watershed	Area (ha)
0-5	Very slight		-	-
5-15	Slight	IV	SW6, SW9, SW10, SW11, SW12 and SW24	18937.8
15-30	Moderate	III	SW5, SW7, SW8, SW19	18418.5
30-50	Sever	II	SW1, SW2,SW3, SW4, SW13, SW14, SW16, SW20, SW21, SW22 and SW23	35375.2
> 50	Very sever	I	SW15, W17 and SW18	7611.5

The effects of LULC change and landscape positions on soil erosion

Raster maps of land use/cover changes and soil loss were superimposed to identify the relationship between these two parameters. The spatiotemporal variability of soil erosion risk is highly impacted by human-induced activities. The analysis results revealed that the rate of soil loss is extremely high in bare lands followed by cultivated fields and open shrub lands. The rate of soil loss from bare land was 511.1% times greater than forest land and soil loss from cultivated field was 105.2% times greater than forest land. This is attributed to the over-exploitation of cultivated fields without applying any soil and water conservation strategies, which is evidenced by field survey work. Frequent cultivation for seed bed preparation disintegrates soil structure and reduces aggregate stability which hastens soil erosion, particularly in slope areas. In addition, cultivated fields and bare lands are exposed to direct rain drop impact contributing to higher soil loss and sediment yield. Poor land use systems and over-exploitation of resources are responsible factors for the expansion of bare lands. On the other hand, the lowest values of soil loss and sediment yield were observed in forest lands. Forest covers reduce rain drop impact on soil particles and the velocity of runoff, thereby significantly reducing soil loss and sediment yield. The relationship between soil erosion risk and the slope of the watershed was detected by, first reclassifying the maps of soil erosion risk and slope of the watershed in to different classes and then overlaying the two raster maps. The highest mean annual soil loss

(60.9 t/ha/yr) was found in the upper and lower parts of the watershed where the slope gradient is greater than 30%. This is due to the effect of the slope of the landscape on the velocity and volume of runoff that greatly impacts soil erosion and sediment transport.

Previous studies conducted in different catchments of Ethiopia showed the impacts of land use and land cover changes on the soil erosion risk and sediment export. The results of the current study are comparable with the findings of previous studies. For instance, a recent study conducted by Aneseye et al. (2020) in Winka watershed, Omo Gibe Basin, Ethiopia indicated that the highest rate of soil erosion is from cultivated land that increased through time from 10.02 t/ha/yr (in 1988) to 43.48 t/ha/yr (in 2018) and the total soil loss change is 176.35 thousand tons over the past 30 years. Similarly, Yesuph and Dagnew (2019) reported the highest rate of soil loss (51 t/ha/yr) from cultivated land in the Beshillo Catchment, Blue Nile Basin, Ethiopia. Likewise, Woldemariam and Harka (2020) indicated an extensive soil loss from cultivated land use (37.06 t/ha/yr) and bare land (15.78 t/ha/yr) from Erer Sub-Basin, Wabi Shebelle Basin, Ethiopia. The study by Gashaw et al. (2019) from Andasa Watershed, Upper Blue Nile Basin, Ethiopia revealed the change in soil loss rate from 35.5 t/ha/yr (in 1985) to 55 t/ha/yr (in 2015) due to an expansion of cultivated land. The highest rate of soil loss from cultivated land could be due to intensive cultivation, and expansion of agricultural fields to steep slopes, and marginal lands with poor management systems (Aneseye et al., 2020; Yesuph and Dagnew, 2019). On the other hand, Nyssen et al. (2009) reported a higher value of soil loss rate from grazing land than cultivated fields in the Tigray region, Ethiopia.

Implications of soil erosion and sediment export to watershed management

Soil erosion and sediment export, which is accelerated by human-induced activities, is a major problem in the Suha watershed, causing soil nutrient depletion, reduction of agricultural productivity, and sediment deposition in downstream reservoirs (Diwediga et al. 2018). The extent and spatial distribution of soil loss and sediment export in response to land-use change and topography vary widely. The results of this study clearly showed that severely degraded area has increased by 32% (25,660 ha) in the past 35 years, and soil and water conservation measures are urgently needed to reverse this condition. The results provide vital information to decision-makers, planners, and development agencies to prioritize sub-watersheds based on soil erosion severity class and select effective soil conservation technologies. Areas with very severe and severe soil erosion classes are recognized as the priority areas for the application of soil and water conservation measures. As soil and water conservation measures require huge investments, limited resources should be allocated to erosion hotspot areas to significantly reduce soil erosion and sediment export.

5.3. Monitoring soil quality in terms of its physical and chemical fertility along the toposequence of the watershed

Effects of land use types and elevation gradient on selected soil physical quality indicators

In this study, the impacts of land use/land cover changes and elevation gradient on selected soil physical and chemical quality indicators were assessed. The relative proportion of soil separate groups determines the soil textural class which significantly affects other soil properties (physical chemical and biological properties). Silt fraction is the most mobile particle by water erosion and the proportions of these fractions are lower in poorly managed and intensively cultivated fields. In the current study, the lowest value of silt fraction was found in the cultivated field followed by grazing land, which is an indicator of the problems of soil erosion in these land use systems. On the other hand, the highest value of clay fraction in the cultivated fields might be attributed to accelerated weathering because of soil disturbance.

Soil texture is an inherent property of soil, and it is the result of soil-forming processes. Its variation could be due to the illuviation of clay particles, removal of clay by soil erosion, biological activities, upward movement of sand fraction by swelling and shrinking phenomena and the combinations of these two or more processes (IUSS Working Group, 2006). Even though it is less influenced by anthropogenic activities in a short period of time (Brejda et al., 2000), in the highland region where there is high rainfall and poor land use systems, smaller particle sizes will be eroded and transported to other locations and resulted in variations in the relative proportions of soil particles and textural classes. This illustrates the non-significant differences of soil separate groups between land use systems in the study area. Similar results were reported by Mulat et al. (2021) who demonstrated that the distribution of soil separate groups didn’t show significant differences among land use systems in the Kersa watershed, Oromia region. Teferi et al. (2016) also reported similar findings from the Jedeb watershed, in northwestern Ethiopia. Soil textural class significantly determines the content of soil water, retention and availability of soil nutrients. For instance, the presence of a higher proportion of sand fraction will negatively affect water holding capacity and nutrient availability (Abera and Assen, 2019; Warra et al., 2015).

Soil aggregate stability is considerably influenced by land use systems and land management strategies. In this study, it was negatively affected in cultivated fields compared to soils under other land use systems. This could be attributed to the depletion of soil organic matter in the cultivated fields and better accumulation of SOM in the forest and grazing lands. These results are in line with the findings of Delelegn et al. (2017) who demonstrated that the value of soil aggregate stability in cultivated land is significantly lower than in other land use systems. Wei et al. (2014) also emphasized that intensive cultivation significantly decreases soil aggregate stability whereas in non-tilled landscapes (forest lands) macro-aggregates of soils are enhanced. Soil organic matter is a key factor to bind soil particles in to aggregates. Due to tillage operation, soil organic matter could be further depleted and hence macro aggregates will be broken down in the cultivated fields. The presence of adequate amount of SOM in the soil enhances soil microbial and fungal biomass that can improve soil aggregate stability (Wu et al., 2015). Mulat et al. (2021) reported higher soil aggregate stability in grazing land use as compared to fallow land and cultivated fields. Peng et al. (2016) also explained the role of soil organic matter in maintaining soil aggregate stability. Similarly, Gebreyesus (2014) examined the negative impacts of soil organic carbon depletion on soil aggregate stability and its consequence on soil degradation. Devine et al. (2014) emphasized that undisturbed ecologies like soils under forest land have better aggregate stability than disturbed ecologies.

Exploring the status of soil water content is essential to devise an irrigation schedule and improve water use efficiency depending on the water requirements of plants. In the present study, the highest amounts of soil water contents (FC and PWP) were obtained in the cultivated field, whereas the lowest values were obtained in the forest land in different elevation gradients. This variation could be attributed to the contents of clay fraction and soil organic matter (SOM). This is also evidenced by the positive and significant correlation of clay with FC ($r = 0.441$, $p < 0.05$) and PWP ($r = 0.932$, $p < 0.01$). Alawamy et al. (2021) reported that the variation in the contents of available water holding capacity (AWHC) among different land use systems might be due to the difference in the contents of clay fraction and soil organic matter

Effects of land use systems and elevation gradient on selected chemical soil quality indicators

Soil pH, SOC, TN, C: N and available phosphorus (Av.P)

The values of soil pH didn't show significant differences among land use systems except location 3 (lower elevation), the soil condition is moderately acidic in all land use systems and locations. Even though the study area is found in the northwestern highlands of Ethiopia, where there is high rainfall and raged topography, the problem of soil acidity is not critical. This might be attributed to the content of high CEC in which basic cations are adsorbed on the surfaces of these colloidal particles. This is also evidenced by positive and significant correlations of pH with CEC (r= 0. 456, $p < 0.01$), PBS (r= 0.425, $p < 0.05$), exchangeable Ca^{2+} (r= 0.486, $p < 0.01$), exchangeable K^{+} (r= 0.491, $p < 0.01$) and exchangeable Mg^{2+} (r= 0.490, $p < 0.01$) (Table 4.21). On the other hand, the lowest values of soil pH in some locations and land uses could be due to the removal of basic cations through soil erosion, leaching, application of acid-forming fertilizers and removal with crop harvest in the cultivated fields. This result is in agreement with the findings of Teferi et al. (2016), who reported a non-significant difference among different land use systems in the Jedeb watershed, northwestern highlands of Ethiopia. On the contrary, Mulat et al. (2021) found significant differences among land use systems in the Kersa watershed, Oromia region. These differences could be due to differences in soil types, climatic conditions and management strategies. Soil pH indicates the toxicity level of aluminum (Al) and the status (deficiency) of soil micro nutrients

Soil organic matter (SOC) is an important soil parameter that highly influences soil physical, chemical and biological properties. It is the key indicator of soil quality and is significantly influenced by anthropogenic activities, vegetation cover and climatic condition (Yu et al., 2020). In this study the lowest values were found in cultivated lands in all elevation gradients and the values are far below the critical level. The content is also low in the grazing land, but optimum in the forest land. Further decomposition of SOM due to continuous and intensive cultivation in the cultivated fields, depletion through soil erosion, complete removal of crop residues from the field (sources of SOM) and low input of organic fertilizers (crop residues and manure) are the major factors for extensive depletion of SOM in the cultivated fields. This leads to a high loss of soil nutrients from these fields as the SOM is the most important soil parameter that determines soil fertility. On the other hand, low soil erosion, continuous litter fall to the soil and the microclimatic conditions under the forest land (low organic matter decomposition) contributes to low soil organic carbon loss and buildup of SOM in the forest land. These results agreed with the findings of Mulat et al. (2021) and Yoseph et al. (2017) who reported higher content of SOM in the grazing land than in cultivated and fallow lands in the Kersa watershed, Eastern and Northern Ethiopia. Feleke et al. (2019) also demonstrated a higher content of SOM in protected grass land than in open grass land due to the frequent turnover of grass and dense root biomass in the protected land use. Similarly, Elias (2019) emphasized the negative impacts of the complete removal of crop residues from cultivated fields and using cow dung as a source of fuel instead of using it as soil amendments to maintain soil nutrients and SOM.

Land use/ land management strategies affect the content of soil TN in the study area. In the cultivated fields, the content of TN is significantly lower than the other land use systems. This variation might be attributed to the amount of soil organic matter in the forest and grazing land, depletion of SOM and leaching of N in the form of nitrate ion from cultivated fields. Taylor et al. (2010) also explained the extent of soil N loss from cultivated fields, but 5 to 100 times higher in the forest land. The amount of SOM present in the soil is a key factor for the presence of TN in adequate amount, which is evidenced by significant and positive correlation of TN with SOM ($r = 0.908$, $p < 0.01$). Similar findings were reported by (Feleke et al., 2019), who demonstrated that the content of TN in the protected grass land was higher than in unprotected grass land because of the variation the in contents of SOM in these land use types.

The C: N ratio determines mineralization or immobilization processes that significantly affect the availability of N to plants. It is an indicator of the rate of SOM decomposition, the amount of SOM held in the soil and the cycling of C and N in a given ecosystem (USDA NRCS, 2020). In general, when the C: N ratio is less than 20: 1, N will be mineralized and when C: N ratio is greater than 30: 1, N will be immobilized. When the C: N ratio is high organic matter decomposition/ mineralization of N could be retarded because of the limited activities of soil microbes. In this study, in all land use systems and locations, the mean values of C: N ratios were less than 20:1, indicating the mineralization process. It also demonstrated the potential impacts of different land use systems and elevation gradients on the C: N ratio. The lowest value was found in the cultivated land in the upper part of the watershed. This could be attributed to further decomposition and removal of SOM as a result of intensive cultivation and soil erosion.

On the other hand, the highest mean value was found in the forest land of the same elevation due to high SOC content. This result is in agreement with Seifu et al. (2020) who explained that the highest value of C: N ratio was found in the grazing land where there is high SOM; whereas the lowest value was recorded in the bare land.

The content of available phosphorus could be influenced by land use system, soil management strategy (application of external inputs) and the amount of soil organic matter. In this study, the highest mean value of available phosphorus was observed in the agricultural land as compared to forest and grazing land use systems. This might be attributed to long-term applications of phosphorus-containing fertilizer (NPSB), which was confirmed during our field survey work and data gathered from farmers and experts. On the other hand, the lowest value in the grazing land might be due to low turnover of grass biomass to the soil because of over grazing. This in turn results in lower soil organic matter and organically bounded phosphorus which could be released to the soil solution through mineralization. In the forest land available phosphorus is higher than grazing land because of better accumulation of SOM. Parallel with this result, Seifu et al. (2020) reported that the highest mean value of available phosphors in the cultivated land as compared to the forest and grazing land use systems in the semiarid watershed, in northwestern Ethiopia. Abera and Assen (2019) also reported similar results from the Wanka watershed, in northwestern Ethiopia. Likewise, De et al. (2022); Tellen and Yerima (2018) explained the higher content of available phosphorus in cultivated land because of continuous application of chemical fertilizers. The high availability of phosphorus is an indication of low p-fixation, which is highly dependent on soil pH (availability decreases in both acidic and alkaline soil conditions). The content of SOM also determines the availability of phosphorus. The findings of previous researches (Feleke et al., 2019) indicated that phosphorus is positively and significantly correlated with SOM and its availability is dependent on the mineralization of SOM. However, in the current study, phosphorus is significantly but negatively correlated with SOM. The reason might be due to the continuous application of phosphorus fertilizer in the cultivated field; conversely, the lowest value of SOM was found in the same field.

Exchangeable bases (Na^+, K^+, Ca^{2+} and Mg^{2+}), CEC and PBS

From this study, the lowest values of exchangeable bases except Mg^{2+} were found in the cultivated land and this might be due to intensive cultivation, leaching of basic cations, soil erosion, low SOM content and removal of cations with crop harvest from this land use type. On the other hand, the highest value of CEC was in the forest land which could be linked with the amount of SOM. Exchangeable K^+ and exchangeable Ca^{2+}were positively and significantly ($r = 0.645$, $p < 0.01$ and $r =0.459$, $p < 0.01$) correlated with SOM; exchangeable Na^+ and Mg^{2+} also showed a positive correlation with SOM ($r = 0.154$ and $r = 0.060$). From the results of exchangeable bases, it was assumed that the exchange sites of the colloidal particles were dominated by divalent cations (Ca^{2+} and Mg^{2+}). According to Bohn et al. (2001), for agricultural productive soils, the order of exchangeable bases should be $Ca^{2+} > Mg^{2+} > K^+ > Na^+$ and any deviation from this could cause cation imbalance for plants. Another important point is the ratio of Ca to Mg cations (Ca: Mg). For most crops the optimal range of Ca: Mg ratio should be between 3: 1 and 4: 1 (Landon, 2014). If the ratio is less than 3:1, it will inhibit the uptake of phosphorus. In the present study, Ca: Mg ratio is less than 3:1; indicating that phosphorus uptake is influenced by this factor.

Cation exchange capacity (CEC) is the potential of soil to hold cations on the exchange site. Marked differences in the mean values of CEC were observed among land use systems and locations. The higher content of CEC in the forest land followed by grazing land might be attributed to the content of SOM. This could be evidenced by positive ($r = 0.31$, $p > 0.05$) correlation of CEC with SOC. In agreement with this result, Abera and Assen (2019) demonstrated that CEC in the natural forest land was higher than in grazing and cultivated lands because of high SOM content in the forest land. The content of CEC also varied along the toposequence of the watershed. The highest value was found at the lower part and this could be due to the large coverage of Vertisol and Leptosol (soil distribution map), with clay minerals having high surface area. This could be substantiated by a positive and significant correlation ($r = 0.548$, $p < 0.01$) of CEC with clay fraction (Table 4.21). In line with this result, Elias E (2019) reported high CEC in Vertisols and Leptosols which are dominated by smectite minerals having high surface area and pH-dependent charges in the cultivated fields of the Ethiopian highlands. The mean values of exchangeable sodium percentage (ESP) were far below its critical level (15) for most agricultural crops in all land uses and locations. This illustrates that the study area is free of sodicity problem, as the area is also found in the north western highlands of Ethiopia. The mean value of PBS in the cultivated field was lower than the values recorded in the forest and grazing lands. But, the mean values of PBS were rated as high in all land use systems. The factors that affect the contents of Exchangeable bases directly affect PBS. Similar findings were reported by Tufa et al. (2019).

Soil quality status under different land use systems

The results of the present study confirmed that soil quality of Suha watershed was declining because of poor managements which were evidenced by the negative values of SQDI for most soil attributes under cultivated and grazing lands. In these land use systems ISS, SOC, TN and exchangeable K were the most deteriorating soil quality indicators. This might be due to soil erosion, leaching, depletion of SOC as a result of intensive cultivation, low return of organic inputs and poor management. On the other hand high positive values of available phosphorus and clay fraction in the cultivated field could be because of continuous application of phosphorus containing chemical fertilizer and cultivation enhances weathering process and hence clay fraction increases. The total SQDI values also showed that soil conditions under cultivated and grazing land use systems have been experiencing progressive deterioration as the result of poor management strategies. These results agreed with the findings of Abera and Assen (2019) who reported declining of soil quality indicators in the disturbed ecosystems (agricultural land and grazing land) in the Wanka watershed, northwestern Ethiopia. Other researchers (Eyayu et al., 2009; Gui et al., 2009) also reported similar findings.

Evaluating the overall status of soil quality through SQI is essential in identifying the soil's potential to give its ecosystem services under different land use systems/ management strategies and landscape positions (Ghosh et al., 2019). In the present study SQI was quantified based on two perspectives (production and soil erosion condition) using four soil characteristics. Significantly lower value of SQI was under cultivated land which could be due to the depletion of soil organic matter, and loss of soil nutrients particularly TN as a result of poor land management. On the other hand, the addition of SOM and less disturbance of the soil resource under the forest resulted in better SQI value. The results of this study were in line with the findings of Feleke et al. (2019), who reported that a higher value of SQI was from protected grass land and a lower value under unprotected grass land in Farta District, northwestern Ethiopia. Bajracharya et al. (2007); Gebreyesus and Vlek (2014) also reported similar findings.

5.4. N and P flows and balances in cereal-based agroecosystems of Suha Watershed, Northwestern Ethiopia

Nutrient inputs and outputs in the two agroecosystems

From our field work and data obtained from district experts and farmers, we have confirmed that mineral fertilizers were the main contributors of nutrient inputs (IN1) for N and P in all farming systems. Cereal crops like wheat and teff received more mineral fertilizers than other vegetables and perennial crops in the study area as also demonstrated by (Haileslassie et al., 2005). The applications of these inputs vary among farming systems which could be explained by the economic capability of purchasing mineral fertilizers. Organic inputs (IN2) were not applied to these fields because of competitive uses for energy sources. Some of the farmers have prepared and applied compost around the homesteads.

In contrast to the addition of nutrients in to the system through inflow mechanisms, nutrients were removed from the system through different outflow pathways. The results of the present study demonstrated that the dominant nutrient outflow mechanisms were crop harvest (OUT1) and residues removal (OUT2). The contents of P were far lower than the contents of N exported through these processes (Table 4.25 and 4.26). The results of this study are in line with the findings of Haileslassie et al. (2005) from the central highlands of Ethiopia; Mulualem et al.(2021) from northwestern Ethiopia. However, considerable variations were observed in different research results which might be attributed to the variation in cropping systems and associated yield.

Nutrient loss through leaching (OUT3) was the main pathway by which N was removed from the farming systems. The results of the present study (30.5 kg N/ha/yr and 25.2 kg N/ha/yr) are comparable with the findings of Haileslassie et al.(2005) who reported a national average value of 17 kg N/ha/y. On the other hand, Mulualem et al. (2021) conducted their study in different land use systems in the Blue Nile Basin and demonstrated that the mean value of N loss from cultivated land was 8.8 kg/ha/yr. Likewise, Stoorvogel and Smaling (1990) also reported the national average leaching loss of N was 3 kg N/ha/yr. The variations in these results could be attributed to the difference in the amount of rainfall, methods of analysis, amount of soluble nutrients in the soil and soil texture (Mulualem et al., 2021; Haileslassie et al., 2005). Moreover, the type and amount of external inputs applied to the field determine the rate of leaching loss.

For example, Carmo et al. (2017) demonstrated that 20% of N-containing fertilizers applied to the crop lands were lost through the leaching process. Tan et al. (2005) also reported that leaching losses of nutrients (particularly N loss) is higher in cultivated fields than in other land-use systems because of the application of easily dissolved fertilizers. The rate of N loss through denitrification depends on the amount of soil moisture content, soil pH, and soil carbon content (Van Bremen et al., 2002). In addition, the amount of external inputs and nutrient uptake by plants significantly determine the rate of N loss by denitrification (Stoorvogel and Smaling, 1990). Keating (1997) indicated that about 25% of the applied N-fertilizer (urea) was lost by the denitrification mechanism. As compared to other nutrient outflow mechanisms, denitrification is the least important for N loss.

Soil erosion is the most important pathway of nutrient removal from the system as compared to other pathways (Van Beek et al., 2016). The amount of nutrients exported through this process in Ethiopia varies across regions due to the difference in soil erodibility, topography, erosivity of rainfall, land cover type and land management (Haileslassie et al., 2005). Cultivated field are more prone to soil erosion and nutrient losses because of poor surface cover and poor soil aggregate stability as a result of repeated ploughing which favors the formation of surface sealing that in turn accelerate soil erosion and nutrient losses (Mulualem et al., 2021). Aggregated results of the present study revealed that 48 kg N/ha/yr and 10.6 kg P/ha/yr were removed with soil loss from the highland and 23 kg N/ha/yr 6.2 kg P/ha/yr from the mid land. Phosphorus removal with sediment was far lower than the N removal which might be attributed to p-fixation (Mulualem et al., 2021; Aticho et al., 2011; Haileslassie et al., 2005). The results of this study are comparable with the findings of previous studies in Ethiopia. For example, Haileslassie et al. (2005) reported the nutrient losses by soil erosion at regional and national scales (using USLE and LAPSUS to estimate soil loss) and demonstrated that the losses of N, P and K from Amhara region were 140.7 kg N/ha/yr, and 29.0 kg P/ha/yr respectively. At the national level, these values were 119.1 kg N/ha/yr, and 24.7 kg P/ha/yr. However, the findings of Mulualem et al. (2021) were far lower than the results of this study.

Soil nutrient balances

Nutrient balance studies provide basic information about the status of soil fertility (where it is improved or depleted).Negative balance (nutrient deficiency) indicates depletion of soil fertility;

whereas positive balances (surplus) indicate the potential/risk of environmental pollution (water, air and soil pollution). In the highlands of Ethiopia, nutrient depletion is becoming severe due to negative nutrient balances which demands actions that reverse this condition (Van Beek et al., 2016). In this study, significant variations of nutrient balances (N, P and K) were observed among socioeconomic groups and between agroecosystems. These variations could be attributed to the difference in yield, farming systems, external inputs used and nutrient use efficiency of each crop. Moreover, variations between agroecologies might be due to the differences in the amount of annual rainfall, landscape position and soil type which are the determining factors of soil erosion.

The results of this study are in agreement with the findings of previous studies conducted in Ethiopia and elsewhere in Sub Sahara African countries (Table 4.28). For instance, the findings of Van Beek et al. (2016) from the highlands of Ethiopia (-24 kg N /ha/yr and +9 kg P /ha/yr) and Mulualem et al. (2021) from the Blue Nile Basin (-50 kg N /ha/yr and -4 kg P /ha/yr) are comparable with the current results. Similarly, Gebresamuel et al. (2020) from northern Ethiopia demonstrated N and P balances of -19.5 to -30.8 kg N/ha/yr and 2.1 to 4.1kg P/ha/yr respectively. The results of N balances were comparable with the findings of other studies in Ethiopia and elsewhere in Sub-Sahara African countries. For instance, the results of Elias (2004) from Southern Ethiopia demonstrated that N balances at farm level were ranged from -24 to -48 kg/ha and from -15 to -37 for the highland and lowland agro-ecological zones respectively. N depletion was higher in the rich farmers in both highland and midland. On the other hand, N and P balance differences between agro-ecological zones revealed that nutrient mining is higher in the highland farming systems than the lowlands (Elias, 2004). These differences could be attributed to higher removal of nutrients with crop harvest (because of higher yield) and soil erosion in the highlands. On the contrary, the results of P balances of this study (-11.9 kg P/ha/yr for the highland and -1.8 kg P/ha/yr for the midland) were different from the results reported from southern Ethiopia (4 to 11 kg P/ha/yr for the highland and from -3 to 10 kg P/ha/yr for the lowland agro-ecological zones respectively). On the other hand, the results of the present study are higher than the findings of Stoorvogel and Smaling (1990), who conducted their research at continental level in Africa and found that N balance was 20 kg N/ha/yr and P balance was 10 kg P/ha/yr. The balances in East Africa were 40 kg N/ha/yr and 20 kg P/ha/yr respectively.

More negative N balances as compared to P balances might be due to:

- Export of large amount of N with crop harvest (product and residues); significant amount of N could be exported with cereal crops which have high N content in their harvested products (Elias, 1998)
- The largest proportion of crop residue was removed from the field for other purposes (mainly for animal feed)
- Absence of the application of organic inputs (which are the sources of organically bounded N) because of the utilization of manures as a source of fuel.
- Removal of a considerable amount of N by soil erosion particularly in the highland area
- Loss of a significant amount of N through leaching and denitrification processes

Phosphorus balances were less negative in both agroecologies and all farming systems. This might be due to the following factors:

- Continuous application of phosphorus-containing mineral fertilizers in both agroecologies and farmlands (even though the rate of application varied)
- Phosphorus is less sensitive to leaching (precipitated by forming compounds in both acidic and alkaline conditions) (Elias, 2002)
- The amount of phosphorus exported with crop harvest (products and residues) was lower than N.
- The loss of P through soil erosion was also relatively smaller as compared to N.

However, these values of P balances are not indicators of sustainable availability of P in the soil. This less negative balance could be due to the continuous application of mineral fertilizers without considering organic sources. Organic inputs (SOM) are also the sources of significant amounts of organic P which can be released in to the soil solution through the mineralization process. The degree of N and P depletions for Sub-Saharan Africa cultivated lands were rated as very severe, severe, moderate and slight (Smaling et al., 1993). These classifications were based on the following rates of losses (Elias et al., 1998): when the depletion of N is > 40 kg N/ha/yr and P loss is > 7 kg P/ha/yr, it is rated as very sever; when N loss is between 20 and 40 N/ha/yr and P loss is between 4 and 7 P/ha/yr, rated as severe; loss of N is between 10 and 20 kg N//4ha/yr and P is between 2 and 4 kg P/ha/yr, rated as moderate. When N loss is < 10 kg

N/ha/yr and P is < 2 kg P/ha/yr, rated as slight. Based on these classifications, the rates of N and P losses in the study area were rated as very sever in the highland (wheat farming system) and sever for N and slight for P in the midland (tef farming system). Lefroy and Wijnhoud (2001) also demonstrate the ratings of K in Sub Saharan Africa countries; (K > 33.2 kg K/ha/yr rated as very high; when K is between 16.6 and 33.2 kg K/ha/yr rated as high; K between 8.3 and 16.6 kg K/ha/yr rated as moderate and K < 8.3 kg K/ha/yr rated as low).

Table 5.3. Comparisons of the results of the present study with previous research findings

Study area	Scale	Nutrient balances (kg/ha/yr)			Source
		N	P_2O_5	K_2O	
Northwestern Ethiopia	Farming system	-39.3 to -77.1	-1.4 to -11.9	-	This study
Central Ethiopia	Regional	-50	-4	-64	Haileslassie et al. (2006)
Ethiopia highlands	District	-24	+9	-7	Van Beek et al. (2016)
Northern Ethiopia	Watershed	-19.5 to -30.8	+2.1 to +4.1	-15.4 to -42.8	Gebresamuel et al. (2020)
Southern Ethiopia	Farm	-92	+5	-49	Elias et al. (2004)
Northern Ethiopia	Watershed	-41	-1	-36	Kiros et al. (2014)
Northern Ethiopia	Regional	-65	-6	-34	Asefa et al. (2003)
Ethiopia	National	-47	-7	-32	Stoorvogel and Smaling (1990)
Jimma zone, Oromia region	Regional	+3	+ 5	-	Aticho et al. (2011)
North western Ethiopia		-20.9 to -61.4	-0.7 to +11	-26.7 to -37.8	Gezie (2019)
Northern Ethiopia (Tigray region)		-17.9 to -26.2	+3.9 to +6.7	-5.2 to + 2.9	Mesfin et al. (2021)

Replacement cost of depleted nutrients from the agroecosystems

Negative nutrient balances are indicators of soil fertility depletion. Moreover, they are highly linked with yield reduction and nutrient replacement costs. The results of the present study are in agreement with Mulualem et al. (2021) who reported that the total replacement costs of N and P for the three agroecologies (highland, midland and lowland of the Upper Blue Nile basin) were US$38, $33 and $26/ha/yr respectively. The replacement cost of K was not considered in their study. Erkosa et al. (2015) conducted their research in the Blue Nile Basin of Ethiopia and reported that significant amounts of soil nutrients (N= 8.6 kg/ha/yr and P= 4.4kg/ha/yr) were exported with sediment and incurred the financial cost of US$210/ha/yr due to soil erosion. Haregeweyn et al. (2008) also reported 68 ETB/ha/yr for the replacement cost of N and P due to soil erosion from the northern highland catchment. Similarly, the findings of Shibabaw and Alemayehu (2015) and Taye et al. (2013) showed that, the replacement cost of N and P was 2273 ETB/ha/yr, and 200 ETB $ha^{-1}yr^{-1}$ respectively. Moreover, Selassie and Belay (2013) found a replacement cost of 6.4 USD $ha^{-1}yr^{-1}$ from non-conserved cultivated lands. FAO (1986) also estimated the replacement cost of N and P which ranged from 29 to 44 ETB/ha/yr for cultivated land. The variation of the results of exported nutrients and associated replacement costs might be attributed to the difference in the farming system, fertility of the soil, amount of mean annual rainfall, amount and type of fertilizer use, and scale of the study. Besides the replacement costs, it is expected that nutrient losses cause yield reduction and off-site effects on dams and reservoirs. When we convert these losses in to monetary bases and estimate the total cost, the result could be far larger than the replacement costs of exported nutrients.

CHAPTER SIX: CONCLUSION AND RECOMMENDATION

6.1. CONCLUSION

Soil degradation in the form of soil erosion by water and nutrient depletion is a major threat to agricultural productivity and food security in the highlands of Ethiopia. Moreover, soil erosion, land use/land cover change, nutrient mining, and poor soil management (particularly removal of residues from the field, insufficient application of external inputs and using animal dung for fuel) are major factors for the deterioration of soil quality. Expansion of agricultural fields to marginal lands and steep slope areas further deteriorates the current soil condition. These problems are not reversed and highly threatened the livelihoods of the rural community. Monitoring the extent and magnitude of land use/cover changes and exploring their associated impacts is critically important to design effective management strategies and to ensure sustainable use of natural resources. Moreover, advancing the knowledge of soil nutrient fluxes and budgets in different agroecosystems could help in developing appropriate policies (strategies) and implementing actions so as to ensure agricultural sustainability. The present study was conducted in the Suha watershed, north western highlands of Ethiopia to analyze land use/ land cover dynamics and evaluate its impacts on the land surface temperature, rate of soil erosion and sediment yield, soil quality change and to quantify soil nutrient flows and balances in different farming systems

The results showed that significant LULC change has been undergone in the study area for the past 35 years (1985 -2019) in which agricultural land and bare land increased by 34.8%, and 373.6% respectively whereas, grazing land and shrub land decreased by 72.1% and 47.6% respectively. Land surface temperature also showed an increasing trend in this time interval and this is obviously linked with land use/cover change. The highest LST values were found on impervious areas of bare land and built-up areas and the lowest value in the vegetation cover areas; which is also evidenced by the negative correlation of NDVI and LST. Expansion of bare land by three and a half fold and built up by one and a half fold in the study period are major factors for increasing trends of LST.

The analysis results of soil erosion and sediment yield also showed that the annual soil loss increased by 1,205,145 t/ha and annual sediment yield by 326,996 t/ha. These results revealed that there is high seasonal and spatial variability of soil loss and sediment yields. Area coverage of sever and very sever soil erosion severity classes were increased significantly by 12,952 ha whereas the areas of very slight and slight soil erosion classes were decreased by 17,743 ha for the entire period. One of the Major drivers of soil erosion risk in the study area was a significant increment in the agricultural and bare lands as it is evidenced by land use/cover change analysis results (1985-2019). Besides, during the field survey, we also confirmed that there is no soil and water conservation measure implemented on soil erosion-prone landscapes that are found on steep slopes of the watershed. Land use /land cover changes (particularly from forest land to agricultural land) had detrimental impacts on soil physical and chemical quality indicators that resulted in the deterioration of soil condition and reduction of agricultural productivity. Soil quality indicators which are sensitive to land use change/management are below their threshold values, indicating that soil condition is deteriorating. Poor management of agricultural lands is the main factor for declining of soil quality indicators which could be explained by the depletion of soil organic carbon (SOC) and nitrogen deficiency. In addition, soil quality degradation index values of most soil quality indicators are highly negative in the cultivated field; which are indicators of poor soil condition. Moreover, aggregated soil quality index value in the agricultural land use system is under the poor class which is an evidence of inappropriate land use system.

The nutrient balance results also showed that nutrient inputs from mineral fertilizers (IN1) are the main sources of inflows whereas nutrient removals by crop harvest (OUT1 and OUT2) and through soil erosion (OUT5) are the major pathways of nutrient outflows from the system. Partial nutrient balances are indicators of the application of sufficient amount of mineral fertilizers. On the other hand, full nutrient balance results indicated that the losses of soil nutrients are strongly linked with crop harvest and soil erosion. Nutrient balances for N and K were negative in both agroecosystems; however, the values were more negative in highland areas due to severe soil erosion as compared to midland farming systems. Replacement costs of exported nutrients would incur additional costs for the small holder farmers. The economic loss could rise when yield loss due to nutrient export is accounted.

6.2. RECOMMENDATION

To reverse the adverse effects of soil erosion and to ensure environmental sustainability, management strategies which include:

- Soil and water conservation technologies in steep slope areas, enhancing afforestation programs and implementation of land use policies of the country should be applied.
- To develop ownership among land owners, community participation is vital in problem identification, planning process, implementation and impact assessment of applied soil and water conservation technologies.
- In addition, soil and site-specific management strategies are required to improve and maintain degraded soil quality indicators so as to improve soil ecosystem services, enhance agricultural productivity and ensure food security. In this regard, application of integrated soil fertility management (ISFM) strategies and soil conservation technologies are suggested as remedial actions.

Concerning nutrient mining, the following soil management strategies that could counter balance negative nutrient balances and improve soil fertility are suggested:

- The first one is the application of recommended mineral fertilizers.
- Second, alternative energy sources should be introduced to reduce the utilization of manure as a source of fuel and to maximize the application of organic inputs in to the field.
- The third strategy is the application of soil and water conservation strategies to halt soil erosion at the watershed scale.

- In addition, planting the same crop for consecutive years is observed as one of the factors that contribute to the decline of soil fertility. Hence, improving this practice, particularly in Vertisols, by adopting Vertisol management strategies is crucial.
- Moreover, linking soil nutrient balances with nutrient reserves (stocks) and considering consecutive cropping seasons are very vital to explore reliable results on the rate of nutrient depletion and sustainability of agroecosystems.

Future considerations/ limitations of this study

1. This research considered dry season periods (January and February) for the analysis of LST; multi-season analysis will contribute additional information to devise management strategies.
2. Even though LULC change is one major factor for LST change considering other parameters affecting LST like, topography and elevation are also important.
3. Soil loss due to gully erosion, stream bank erosion, and landslides was not considered in this study since RUSLE is incapable of analyzing soil loss from these soil erosion processes. Therefore, in order to fully comprehend the risk of soil erosion in this area, additional approaches should be incorporated.
4. The impacts of soil erosion on soil nutrient export and related replacement costs, agricultural productivity and food security should be investigated.
5. Biological soil quality indicators were not considered in this study due to financial limitations; these parameters play a great role in improving soil conditions. Hence, evaluating these parameters is required to develop sound full management strategies.

7. REFERENCES

Abate, S. 2011b. Evaluating the land use and land cover dynamics in Borena Woreda of South Wollo Highlands, Ethiopia. *Journal of Sustainable Development in Africa*, *13*(1), pp.87-107.

Abera, W. and Assen, M. 2019. Dynamics of selected soil quality indicators in response to land use/cover and elevation variations in Wanka watershed, northwestern Ethiopian highlands. *Ekológia*, *38*(2), pp.126-139.

Addis, H.K., Strohmeier, S., Ziadat, F., Melaku, N.D. and Klik, A. 2016. Modeling streamflow and sediment using SWAT in Ethiopian Highlands. *International Journal of Agricultural and Biological Engineering*, *9*(5), pp.51-66.

Adimassu, Z., Kessler, A. and Hengsdijk, H. 2012. Exploring determinants of farmers' investments in land management in the Central Rift Valley of Ethiopia. *Applied Geography*, *35*(1-2), pp.191-198.

Aga, A.O., Melesse, A.M. and Chane, B. 2019. Estimating the Sediment Flux and Budget for a Data Limited Rift Valley Lake in Ethiopia. *Hydrology*, *7*(1), p.3.

AGRA 2014.

AGRA Africa Agriculture Status Report (2014) Climate change and smallholder agriculture in sub-Saharan Africa.

Ahmad, A., Hashim, U.K.M., Mohd, O., Abdullah, M.M., Sakidin, H., Rasib, A.W. and Sufahani, S.F. 2018. Comparative analysis of support vector machine, maximum likelihood and neural network classification on multispectral remote sensing data. *International Journal of Advanced Computer Science and Applications*, *9* (9), pp.529-537.

Ahmed, A.A. and Ismail, U.H.A.E. 2008. Sediment in the Nile River system. *Consultancy Study requested by UNESCO*.

Ahmed, O.H., Aminuddin, H. and Husni, M.H.A. 2006. Reducing ammonia loss from urea and improving soil-exchangeable ammonium retention through mixing triple superphosphate, humic acid, and zeolite. *Soil Use and Management*, *22*(3), pp.315-319.

Alam, A., Bhat, M.S. and Maheen, M. 2020. Using Landsat satellite data for assessing the land use and land cover change in Kashmir valley. *GeoJournal*, *85*(6), pp.1529-1543.

Alarima, C.I., Annan-Afful, E., Obalum, S.E., Awotunde, J.M., Masunaga, T., Igwe, C.A. and Wakatsuki, T. 2020. Comparative assessment of temporal changes in soil degradation under four contrasting land-use options along a tropical toposequence. *Land Degradation & Development*, *31*(4), pp.439-450.

Alawamy, J.S., Balasundram, S.K. and Boon Sung, C.T. 2020. Detecting and analyzing land use and land cover changes in the region of Al-Jabal Al-Akhdar, Libya using time-series landsat data from 1985 to 2017. *Sustainability*, *12*(11), p.4490.

Alawamy, J.S., Balasundram, S.K., Mohd. Hanif, A.H. and Boon Sung, C.T. 2021. Response of Potential Indicators of Soil Quality to Land-Use and Land-Cover Change under a

Mediterranean Climate in the Region of Al-Jabal Al-Akhdar, Libya. *Sustainability*, *14*(1), p.162.

Alemu, B., Garedew, E., Eshetu, Z. and Kassa, H. 2015. Land Use and Land Cover Changes and associated Driving Forces in North Western Lowlands of Ethiopia. Int. Res. J. Agric. Sci. and Soil Sci. 5(1):28-44

Alemu, M.M. 2019. Analysis of Spatio-temporal Land Surface Temperature and Normalized Difference Vegetation Index Changes in the Andassa Watershed, Blue Nile Basin, Ethiopia. *Journal of Resources and Ecology*, *10*(1), pp.77-85.

Amede, T. 2003. Opportunities and challenges in reversing land degradation: The regional experience.

Andrews, S.S., Karlen, D.L. and Cambardella, C.A. 2004. The soil management assessment framework: A quantitative soil quality evaluation method. *Soil Science Society of America Journal*, *68*(6), pp.1945-1962.

Andrews, S.S., Karlen, D.L. and Mitchell, J.P. 2002. A comparison of soil quality indexing methods for vegetable production systems in Northern California. *Agriculture, ecosystems & environment*, *90*(1), pp.25-45.

Aneseyee, A. B., Elias, E., Soromessa, T. and Feyisa, G. L. 2020.Land use/land cover change effect on soil erosion and sediment delivery in the Winike watershed, Omo Gibe Basin, Ethiopia.*Science of the Total Environment*, 728 (2020), 138776.

Anon. 1954. Diagnosis and improvement of saline alkali soils. USDA Agri. Handbook No. 60. 160 p.

Aredehey, G., Berhe Zenebe, G. and Gebremedhn, A. 2019. Land use impacts on physicochemical and microbial soil properties across the agricultural landscapes of Debrekidan, EasternTigray, Ethiopia. *Cogent Food & Agriculture*, *5*(1), p.1708683.

Arekhi, S., Niazi, Y. and Kalteh, A.M. 2012. Soil erosion and sediment yield modeling using RS and GIS techniques: a case study, Iran. *Arabian Journal of Geosciences*, *5*(2), pp.285-296.

Argent, R.M., Voinov, A., Maxwell, T., Cuddy, S.M., Rahman, J.M., Seaton, S., Vertessy, R.A. and Braddock, R.D., 2005. Comparing modelling frameworks–a workshop approach. *Environmental Modelling & Software*, *21*(7), pp.895-910.

Artis, D. A. and Carnahan, W. H. 1982."Survey of emissivity variability in thermography of urban areas. *Remote sensing of Environment*, *12*(4), pp.313-329.

Assefa, A. and van Keulen, H., 2005. Soil nutrient dynamics under alternative farm management practices in integrated crop-livestock systems in the Northern Highlands of Ethiopia: A simulation study. *Farm management in mixed crop-livestock systems in the Northern Highlands of Ethiopia*, p.143.

Asensio, V., Guala, S.D., Vega, F.A. and Covelo, E.F. 2013. A soil quality index for reclaimed mine soils. *Environmental toxicology and chemistry*, *32*(10), pp.2240-2248.

Athick, A.M.A. and Shankar, K. 2019. Data on land use and land cover changes in AdamaWereda, Ethiopia, on ETM+, TM and OLI-TIRS landsat sensor using PCC and CDM techniques. *Data in brief*, *24*.

Aticho, A., Elias, E. and Diels, J. 2011. Comparative analysis of soil nutrient balance at farm level: a case study in Jimma Zone, Ethiopia. *International Journal of Soil Science*, *6*(4), p.259.

Atoma, H., Suryabhagavan, K.V. and Balakrishnan, M. 2020. Soil erosion assessment using RUSLE model and GIS in Huluka watershed, Central Ethiopia. *Sustainable Water Resources Management*, *6*(1), pp.1-17.

Auerswald, K., Kainz, M., Schröder, D. and Martin, W. 1992. Comparison of German and Swiss rainfall simulators. Experimental setup. Utility, labour demands and costs. *Zeitschrift für Pflanzenernährung und Bodenkunde*, *155*(1), pp.1-11.

Avdan, U. and Jovanovska G. 2006. "Algorithm for automated mapping of land surface temperature using LANDSAT 8 satellite data." *Journal of sensors*.

Awuh, M.E., Japhets, P.O., Officha, M.C., Okolie, A.O. and Enete, I.C. 2019. A Correlation Analysis of the Relationship between Land Use and Land Cover/Land Surface Temperature in Abuja Municipal, FCT, Nigeria. *Journal of Geographic Information System*, *11*(01), p.44.

Ayalew, G. 2015. A geographic information system based soil loss and sediment estimation in Zingin watershed for conservation planning, highlands of Ethiopia. *World Appl Sci J*, *33*(1), pp.69-79.

Bahr, E., Chamba-Zaragocin, D., Fierro-Jaramillo, N., Witt, A. and Makeschin, F., 2015. Modeling of soil nutrient balances, flows and stocks revealed effects of management on soil fertility in south Ecuadorian smallholder farming systems. *Nutrient cycling in agroecosystems*, *101*, pp.55-82.

Bajracharya, R.M., Sitaula, B.K., Sharma, S. and Jeng, A. 2007. Soil quality in the Nepalese context–An analytical review. *International Journal of Ecology and Environmental Sciences*, *33*(2-3), pp.143-158.

Balew, A. and Korme, T. 2020. Monitoring land surface temperature in Bahir Dar city and its surrounding using Landsat images. *The Egyptian Journal of Remote Sensing and Space Science*, *23*(3), pp.371-386.

Bantider, A., Hurni, H. and Zeleke, G. 2011. Responses of rural households to the impacts of population and land-use changes along the Eastern Escarpment of Wello, Ethiopia. *Norsk Geografisk Tidsskrift-Norwegian Journal of Geography*, *65*(1), pp.42-53.

Barlow, J., Lennox, G.D., Ferreira, J., Berenguer, E., Lees, A.C., Nally, R.M., Thomson, J.R., Ferraz, S.F.D.B., Louzada, J., Oliveira, V.H.F. and Parry, L., 2016. Anthropogenic disturbance in tropical forests can double biodiversity loss from deforestation. *Nature*, *535*(7610), pp.144-147.

Bastida, F., Zsolnay, A., Hernández, T. and García, C., 2008. Past, present and future of soil quality indices: a biological perspective. *Geoderma*, *147*(3-4), pp.159-171.

Behera, M., Sena, D.R., Mandal, U., Kashyap, P.S. and Dash, S.S. 2020. Integrated GIS-based RUSLE approach for quantification of potential soil erosion under future climate change scenarios. *Environmental Monitoring and Assessment*, *192*(11), pp.1-18.

Bekele, B. and Gemi, Y. 2021. Soil erosion risk and sediment yield assessment with universal soil loss equation and GIS: in Dijo watershed, Rift valley Basin of Ethiopia. *Modeling Earth Systems and Environment*, *7*(1), pp.273-291.

Belay, T. and Mengistu, D.A. 2019. Land use and land cover dynamics and drivers in the Muga watershed, Upper Blue Nile basin, Ethiopia. *Remote Sensing Applications: Society and Environment*, *15*, p.100249.

Belayneh, M., Yirgu, T. and Tsegaye, D. 2019. Potential soil erosion estimation and area prioritization for better conservation planning in Gumara watershed using RUSLE and GIS techniques'. *Environmental Systems Research*, *8*(1), pp.1-17.

Belenok, V., Noszczyk, T., Hebryn-Baidy, L. and Kryachok, S. 2021. Investigating anthropogenically transformed landscapes with remote sensing, *Remote Sensing Applications: Society and Environment* 24 (2021) 100635.

Benavidez, R., Bethanna, J., Deborah, M. and Kevin, N. 2018. "A review of the Revised Universal Soil Loss Equation ((RUSLE): With a view to increasing its global applicability and improving soil loss estimates." *Hydrology and Earth System Sciences*, 22 (11), pp. 6059-6086.

Berihun, M.L., Tsunekawa, A., Haregeweyn, N., Dile, Y.T., Tsubo, M., Fenta, A.A., Meshesha, D.T., Ebabu, K., Sultan, D. and Srinivasan, R. 2020. Evaluating runoff and sediment responses to soil and water conservation practices by employing alternative modeling approaches. *Science of the Total Environment*, *747*, p.141118.

Bewket, W. and Stroosnijder, L. 2003. Effects of agroecological land use succession on soil properties in Chemoga watershed, Blue Nile basin, Ethiopia. *Geoderma*, *111*(1-2), pp.85-98.

Bewket, W. and Teferi, E. 2009. Assessment of soil erosion hazard and prioritization for treatment at the watershed level: case study in the Chemoga watershed, Blue Nile Basin, Ethiopia. *Land Degradation & Development*, *20*(6), pp.609-622.

Bindraban, P.S., Stoorvogel, J.J., Jansen, D.M., Vlaming, J. and Groot, J.J.R., 2000. Land quality indicators for sustainable land management: proposed method for yield gap and soil nutrient balance. *Agriculture, Ecosystems & Environment*, *81*(2), pp.103-112.

Bing, H., Wu, Y., Liu, E. and Yang, X., 2013. Assessment of heavy metal enrichment and its human impact in lacustrine sediments from four lakes in the mid-low reaches of the Yangtze River, China. *Journal of Environmental Sciences*, *25*(7), pp.1300-1309.

Birhanu, A., Masih, I., Van Der Zaag, P., Nyssen, J. and Cai, X. 2019. Impacts of land use and land cover changes on hydrology of the Gumara catchment, Ethiopia. *Physics and Chemistry of the Earth, Parts a/b/c*, *112*, pp.165-174.

Blanco-Canqui, H. and Lal, R. 2008. Climate change and soil erosion risks. *Principles of Soil Conservation and Management*, pp.513-536.

Bouyoucus, G. 1962. Hydrometer method improvement for making particle size analysis of soils. Agron J 54:179–186.

Braimoh, A.K. and Vlek, P.L.G., 2006. Soil quality and other factors influencing maize yield in northern Ghana. *Soil Use and Management*, *22*(2), pp.165-171.

Brejda, J. J., Douglas, L., Karlen, J., Smith, L. and Deborah L. 2000. "Identification of regional soil quality factors and indicators II. Northern Mississippi Loess Hills and Palouse Prairie." *Soil Science Society of America Journal* 64(6), pp 2125-2135.

Bremner, J.M. and Mulvaney, C.S. 1982. Total nitrogen In: Page, AL, RH Miller, and DR Keeney (Eds). Methods of Soil Analysis. Part 2. *Amer. Soc. Agron. Madison, WI USA*, pp.595-624.

Brhane, G. and Mekonnen, K. 2009. Estimating soil loss using Universal Soil Loss Equation (USLE) for soil conservation planning at Medego watershed, Northern Ethiopia. *Journal of American Science*, *5*(1), pp.58-69.

Brovkin, V., Boysen, L., Arora, V.K., Boisier, J.P., Cadule, P., Chini, L., Claussen, M., Friedlingstein, P., Gayler, V., Van Den Hurk, B.J.J.M. and Hurtt, G.C. 2013. Effect of anthropogenic land-use and land-cover changes on climate and land carbon storage in CMIP5 projections for the twenty-first century. *Journal of Climate*, *26*(18), pp.6859-6881.

Brussaard, L. 2012. Ecosystem services provided by the soil biota. *Soil ecology and ecosystem services*, (1995).

Bünemann, E.K., Bongiorno, G., Bai, Z., Creamer, R.E., De Deyn, G., De Goede, R., Fleskens, L., Geissen, V., Kuyper, T.W., Mäder, P. and Pulleman, M. 2018. Soil quality–A critical review. *Soil Biology and Biochemistry*, *120*, pp.105-125.

Carmo, M., García-Ruiz, R., Ferreira, M.I. and Domingos, T. 2017. The NPK soil nutrient balance of Portuguese cropland in the 1950s: The transition from organic to chemical fertilization. *Scientific Reports*, *7*(1), pp.1-14.

Chander, G., Markham, B.L. and Helder, D.L. 2009. Summary of current radiometric calibration coefficients for Landsat MSS, TM, ETM+, and EO-1 ALI sensors. *Remote sensing of environment*, *113*(5), pp.893-903.

Chander, G. and Markham, B. 2003. Revised Landsat-5 TM radiometric calibration procedures and postcalibration dynamic ranges. *IEEE Transactions on geoscience and remote sensing*, *41*(11), pp.2674-2677.

Chandramohan, T., Venkatesh, B. and Balchand, A.N. 2015. Evaluation of three soil erosion models for small watersheds. *Aquatic Procedia*, *4*, pp.1227-1234.

Chapman, H. D. 1965. "Cation-exchange capacity." *Methods of soil analysis: Part 2 Chemical and microbiological properties* 9 (1965): 891-901.

Chowdhury, M., Hasan, M.E. and Abdullah-Al-Mamun, M.M. 2020. Land use/land cover change assessment of Halda watershed using remote sensing and GIS. *The Egyptian Journal of Remote Sensing and Space Science*, *23*(1), pp.63-75.

Cobo, J.G., Dercon, G. and Cadisch, G. 2010. Nutrient balances in African land use systems across different spatial scales: A review of approaches, challenges, and progress. *Agriculture, ecosystems & environment*, *136*(1-2), pp.1-15.

Congalton, R.G. 2005. "Thematic and positional accuracy assessment of digital remotely sensed data." *In: McRoberts, Ronald E.; Reams, Gregory A.; Van Deusen, Paul C.; McWilliams, William H., eds. Proceedings of the seventh annual forest inventory and analysis symposium;*

October 3-6, 2005; Portland, ME. Gen. Tech. Rep. WO-77. Washington, DC: US Department of Agriculture, Forest Service: 149-154., vol. 77.

Congalton, R.G., 2009. Accuracy and error analysis of global and local maps: Lessons learned and future considerations. *Remote Sensing of Global Croplands for Food Security*, *441*, pp.47-55.

Congedo, L., 2021. Semi-Automatic Classification Plugin: A Python tool for the download and processing of remote sensing images in QGIS. *Journal of Open Source Software*, *6*(64), p.3172.

Coppus, R. and Imeson, A.C. 2002. Extreme events controlling erosion and sediment transport in a semi-arid sub-Andean valley. *Earth Surface Processes and Landforms: The Journal of the British Geomorphological Research Group*, *27*(13), pp.1365-1375.

Corstanje, R., Mercer, T.G., Rickson, J.R., Deeks, L.K., Newell-Price, P., Holman, I., Kechavarsi, C. and Waine, T.W. 2017. Physical soil quality indicators for monitoring British soils. *Solid Earth*, *8*(5), pp.1003-1016.

Dagnachew, M., Moges, A. and Kassa, A.K. 2019. Effects of land uses on soil quality indicators: The case of Geshy subcatchment, Gojeb River Catchment, Ethiopia. *Applied and Environmental Soil Science*.

Damtea, W., Kim, D. and Im, S. 2020. Spatiotemporal analysis of land cover changes in the chemoga basin, Ethiopia, using Landsat and google earth images. *Sustainability*, *12*(9), p.3607.

Das, S. and Angadi, D.P. 2020. Land use-land cover (LULC) transformation and its relation with land surface temperature changes: A case study of Barrackpore Subdivision, West Bengal, India. *Remote Sensing Applications: Society and Environment*, *19*, p.100322.

Davari, M., Gholami, L., Nabiollahi, K., Homaee, M. and Jafari, H.J. 2020. Deforestation and cultivation of sparse forest impacts on soil quality: case study in West Iran, Baneh. *Soil and Tillage Research*, *198*, p.104504.

Debie, E., Singh, K.N. and Belay, M. 2019. Effect of conservation structures on curbing rill erosion in micro-watersheds, northwest Ethiopia. *International soil and water conservation research*, *7*(3), pp.239-247.

Degife, A., Worku, H. and Gizaw, S. 2021. Environmental implications of soil erosion and sediment yield in Lake Hawassa watershed, south-central Ethiopia. *Environmental Systems Research*, *10*(1), pp.1-24.

Degife, A., Worku, H., Gizaw, S. and Legesse, A. 2019. Land use land cover dynamics, its drivers and environmental implications in Lake Hawassa Watershed of Ethiopia. *Remote sensing applications: society and environment*, *14*, pp.178-190.

De Paul, V. 2019. Integrating management information with soil quality dynamics to monitor agricultural productivity. *Science of the Total Environment*, *651*, pp.2036-2043.

De Paul, V. and Lal, R. 2016. Towards a standard technique for soil quality assessment. *Geoderma*, *265*, pp.96-102.

De, P., Deb, S., Deb, D., Chakraborty, S., Santra, P., Dutta, P., Hoque, A. and Choudhury, A. 2022. Soil quality under different land uses in eastern India: Evaluation by using soil indicators and quality index. *Plos one*, *17*(9), p.e0275062.

Delelegn, Y.T., Purahong, W., Blazevic, A., Yitaferu, B., Wubet, T., Göransson, H. and Godbold, D.L. 2017. Changes in land use alter soil quality and aggregate stability in the highlands of northern Ethiopia. *Scientific Reports*, *7*(1), pp.1-12.

Dessie, G. and Johan, K. 2007. "Pattern and magnitude of deforestation in the South Central Rift Valley Region of Ethiopia." *Mountain research and development,* 27(2), pp.162-168.

Devi, R., Tesfahune, E., Legesse, W., Deboch, B. and Beyene, A. 2008. Assessment of siltation and nutrient enrichment of Gilgel Gibe dam, Southwest Ethiopia. *Bioresource Technology*, *99*(5), pp.975-979.

Devine, S., Markewitz, D., Hendrix, P. and Coleman, D. 2014. Soil aggregates and associated organic matter under conventional tillage, no-tillage, and forest succession after three decades. *PloS one*, *9*(1), p.e84988.

Diao, X., Peter, H. and James, T. 2010. "The role of agriculture in African development." *World development,* 38(10), pp.1375-1383.

Dibaba, W.T., Demissie, T.A. and Miegel, K. 2021. Prioritization of Sub-Watersheds to Sediment Yield and Evaluation of Best Management Practices in Highland Ethiopia, Finchaa Catchment. *Land*, *10*(6), p.650.

Dinka, M.O. and Chaka, D.D. 2019. Analysis of land use/land cover change in Adei watershed, Central Highlands of Ethiopia. *Journal of Water and Land Development*.

Diwediga, B., Le, Q.B., Agodzo, S.K., Tamene, L.D. and Wala, K. 2018. Modelling soil erosion response to sustainable landscape management scenarios in the Mo River Basin (Togo, West Africa). *Science of the Total Environment*, *625*, pp.1309-1320.

Dobereiner, J., Urquiaga, S. and Boddey, R.M. 1996. Alternatives for nitrogen nutrition of crops in tropical agriculture. In *Nitrogen Economy in Tropical Soils: Proceedings of the International Symposium on Nitrogen Economy in Tropical Soils, held in Trinidad, WI, January 9–14, 1994* (pp. 338-346), Netherlands.

Easton, Z.M., Fuka, D.R., Walter, M.T., Cowan, D.M., Schneiderman, E.M. and Steenhuis, T.S. 2008. Re-conceptualizing the soil and water assessment tool (SWAT) model to predict runoff from variable source areas. *Journal of hydrology*, *348*(3-4), pp.279-291.

Elias E. 2004. Nutrient flow analysis: A case study from southern Ethiopia. *Ethiopian journal of Natural Resources,* 6(1): 1-23.

Elias, E. 2002. Farmers perceptions of soil fertility change and management. SOS-Sahel and Institute for Sustainable Development.146p.

Elias, E. 2019. Selected chemical properties of agricultural soils in the Ethiopian highlands: a rapid assessment. *South African Journal of Plant and Soil*, *36*(2), pp.153-156.

Elias, E., Morse, S. and Belshaw, D.G.R. 1998. Nitrogen and phosphorus balances of Kindo Koisha farms in southern Ethiopia. *Agriculture, ecosystems and environment*, *71*(1-3), pp.93-113.

Eniolorunda, N.B. and Bello, A.G. 2016. Forest Cover Change Assessment of Tangaza Forest Reserve, North-West. *Ife Research Publications in Geography*, *10*(1), pp.66-67.

Erencin, Z., Shresta, D.P. and Krol, I.B. 2000. C-factor mapping using remote sensing and GIS. Case study Lom SakLom Kao Thail Geogr Inst Justus-Liebig-Univ Giess Intern Inst Aerosp Surv Earth SciITC Enschede Netherland.

Erkossa, T., Wudneh, A., Desalegn, B. and Taye, G. 2015. Linking soil erosion to on-site financial cost: lessons from watersheds in the Blue Nile Basin. *Solid Earth*, *6*(2), pp.765-774.

Esa, E., Assen, M. and Legass, A. 2018. Implications of land use/cover dynamics on soil erosion potential of agricultural watershed, northwestern highlands of Ethiopia. *Environmental Systems Research*, *7*(1), pp.1-14.

Eshetu, Z., Simane, B., Tebeje, G., Negatu, W., Amsalu, A., Berhanu, A., Bird, N., Welham, B. and Trujillo, N.C. 2014. Climate finance in Ethiopia. *Overseas Development Institute, London and Climate Science Centre, Addis Ababa*.

Estefan, G., Sommer, R. and Ryan J. 2013. *Methods of soil, plant, and water analysis : A manual for the West Asia and North Africa region*. Beirut: ICARDA.

EthioSIS (Ethiopia Soil Information System). 2014. "Soil Fertility Status and Fertilizer Recommendation Atlas for Tigray Regional State." p76.

Eva, H.D., Brink, A. and Simonetti, D. 2006. Monitoring land cover dynamics in sub-Saharan Africa. *Institute for Environmental and Sustainability, Tech. Rep. EUR*, *22498*.

Eyayu, M., Heluf, G., Tekalign, M. and Mohammed, A. 2009. Effects of land-use change on selected soil properties in the Tera Gedam Catchment and adjacent agroecosystems, north-west Ethiopia. *Ethiopian Journal of Natural Resources*, *11*(1), pp.35-62.

Ezeaku, P.I. 2015. Evaluation of agro-ecological approach to soil quality assessment for sustainable land use and management systems. *Scientific Research and Essays*, *10*(15), pp.501-512.

Feleke, T.G., Sharma, P.D. and Selfeko, D.T. 2019. Assessing soil quality of Abargay Rangeland in Farta District, Amhara Regional State, Ethiopia. *Journal of Soil Science and Environmental Management*, *10*(3), pp.46-57.

Fenta, A.A., Tsunekawa, A., Haregeweyn, N., Tsubo, M., Yasuda, H., Kawai, T., Ebabu, K., Berihun, M.L., Belay, A.S. and Sultan, D. 2021. Agroecology-based soil erosion assessment for better conservation planning in Ethiopian river basins. *Environmental Research*, *195*, p.110786.

Fentie, S.F., Jembere, K., Fekadu, E. and Wasie, D. 2020. Land use and land cover dynamics and properties of soils under different land uses in the tejibara watershed, Ethiopia. *Scientific World Journal*, *2020*.

Ferreira, V., Panagopoulos, T., Cakula, A., Andrade, R. and Arvela, A. 2015. Predicting soil erosion after land use changes for irrigating agriculture in a large reservoir of southern Portugal. *Agriculture and Agricultural Science Procedia*, *4*, pp.40-49.

Food and Agriculture Organization (FAO). 1986. Highlands reclamation study. Ethiopia. Final report, Vol. 1. FAO, Rome, pp. 334.

Food and Agriculture Organization (FAO). 2003. Scaling Soil Nutrient Balances. FAO, Rome.

Food and Agriculture Organization (FAO). 2006. Guide for soil description, 4th edn. FAO, Rome

Food and Agriculture Organization (FAO). 2015. Intergovernmental Technical Panel on Soils (ITPS). Status of the World's Soil Resources (SWSR) Main Report; FAO and ITPS: Rome, Italy.

Food and Agriculture Organization (FAO). 2017. Soil erosion: the greatest challenge to sustainable soil management. Rome. p,100.

Foody, G.M. and Mathur, A. 2004. Toward intelligent training of supervised image classifications: directing training data acquisition for SVM classification. *Remote Sensing of Environment*, *93*(1-2), pp.107-117.

Gachene, C.K.K., 1995. Evaluation and mapping of soil erosion susceptibility: an example from Kenya. *Soil Use and Management*, *11*(1), pp.1-4.

Gallagher, P., Baumes, H. (2012). Biomass supply from corn residues: estimates and critical reviews of procedures. Agricultural economic report 847, USDA.

García-Ruiz, J.M., Beguería, S., Nadal-Romero, E., González-Hidalgo, J.C., Lana-Renault, N. and Sanjuán, Y. 2015. A meta-analysis of soil erosion rates across the world. *Geomorphology*, *239*, pp.160-173.

Garrigues, E., Corson, M.S., Angers, D.A., van der Werf, H.M. and Walter, C. 2012. Soil quality in Life Cycle Assessment: Towards development of an indicator. *Ecological indicators*, *18*, pp.434-442.

Gashaw, T., Bantider, A., Zeleke, G., Alamirew, T., Jemberu, W., Worqlul, A.W., Dile, Y.T., Bewket, W., Meshesha, D.T., Adem, A.A. and Addisu, S. 2021. Evaluating InVEST model for estimating soil loss and sediment export in data scarce regions of the Abbay (Upper Blue Nile) Basin: Implications for land managers. *Environmental Challenges*, *5*, p.100381.

Gashaw, T., Tulu, T., Argaw, M. and Worqlul, A.W. 2019. Modeling the impacts of land use–land cover changes on soil erosion and sediment yield in the Andassa watershed, upper Blue Nile basin, Ethiopia. *Environmental Earth Sciences*, *78*(24), pp.1-22.

Gashaw, T., Tulu, T., Argaw, M. and Worqlul, A.W. 2017. Evaluation and prediction of land use/land cover changes in the Andassa watershed, Blue Nile Basin, Ethiopia. *Environmental Systems Research*, *6*(1), pp.1-15.

Gashaw, T., Bantider, A. and Mahari, A. 2014. Evaluations of land use/land cover changes and land degradation in Dera District, Ethiopia: GIS and remote sensing based analysis. *International Journal of Scientific Research in Environmental Sciences*, *2*(6), p.199.

Gebremedhin, K., Mitiku, H. and Girmay, G. 2014. Assessing the input and output flows and nutrients balance analysis at catchment level in Northern Ethiopia. *Journal of Soil Science and Environmental Management*, *5*(1), pp.1-12.

Gebremicael, T.G., Mohamed, Y.A., Betrie, G.D., Van der Zaag, P. and Teferi, E. 2013. Trend analysis of runoff and sediment fluxes in the Upper Blue Nile basin: A combined analysis of statistical tests, physically-based models and landuse maps. *Journal of Hydrology*, *482*, pp.57-68.

Gebresamuel, G., Opazo-Slazar, D., Collar-Nunez, G., van Beek, C., Elias, E. and Okolo, C.C. 2020. Nutrient Balance of Farming Systems in Tigray, Northern Ethiopia. *Journal of Soil Science and Plant Nutrition*.

Gebreselassie, S., Kirui, O.K. and Mirzabaev, A. 2016. Economics of land degradation and improvement in Ethiopia. *Economics of land degradation and improvement–a global assessment for sustainable development*, pp.401-430.

Gebreyesus, B. and Vlek, P. L. G. 2014. Assessing sediment-nutrient export rate and soil degradation in Mai-Negus catchment, Northern Ethiopia. International Scholarly Research Notices, *2013*.

Gebreyesus, B. 2014. Soil quality assessment strategies for evaluating soil degradation in Northern Ethiopia," Applied and Environmental Soil Science, 14, pp.64-78.

Gedefaw, A.A., Atzberger, C., Bauer, T., Agegnehu, S.K. and Mansberger, R. 2020. Analysis of land cover change detection in Gozamin district, Ethiopia: From remote sensing and DPSIR perspectives. *Sustainability*, *12*(11), p.4534.

Gelagay, H.S. and Minale, A.S. 2016. Soil loss estimation using GIS and Remote sensing techniques: A case of Koga watershed, Northwestern Ethiopia. *International Soil and Water Conservation Research*, *4*(2), pp.126-136.

Gémes, O., Tobak, Z. and Van Leeuwen, B., 2016. Satellite based analysis of surface urban heat island intensity. *Journal of environmental geography*, *9*(1-2), pp.23-30.

Gezie, M. 2019. Nutrient balance in small catchment of the upland areas of the Gumara River, northwestern Ethiopia. MSc thesis, Bahir Dar University, Ethiopia.

Ghosh, S., Deb, S., Ow, L.F., Deb, D. and Yusof, M.L. 2019. Soil characteristics in an exhumed cemetery land in Central Singapore. *Environmental monitoring and assessment*, *191*(3), pp.1-13.

Gui, D., Lei, J., Mu, G. and Zeng, F. 2009. Effects of different management intensities on soil quality of farmland during oasis development in southern Tarim Basin, Xinjiang, China. *International Journal of Sustainable Development & World Ecology*, *16*(4), pp.295-301.

Haileslassie, A., Priess, J.A., Veldkamp, E. and Lesschen, J.P. 2006. Smallholders' soil fertility management in the Central Highlands of Ethiopia: implications for nutrient stocks, balances and sustainability of agroecosystems. *Nutrient Cycling in Agroecosystems*, *75*, pp.135-146.

Haileslassie, A., Priess, J., Veldkamp, E., Teketay, D. and Lesschen, J. P.2005. Assessment of soil nutrient depletion and its spatial variability on smallholders' mixed farming systems in

Ethiopia using partial versus full nutrient balances. *Agriculture, Ecosystems and Environment*, 108, pp.1–16.

Haregeweyn, N., Poesen, J., Deckers, J., Nyssen, J., Haile, M., Govers, G., Verstraeten, G. and Moeyersons, J. 2008. Sediment-bound nutrient export from micro-dam catchments in Northern Ethiopia. *Land degradation & development*, *19*(2), pp.136-152.

Haregeweyn, N., Tsunekawa, A., Nyssen, J., Poesen, J., Tsubo, M., Meshesha, D.T., Schütt, B., Adgo, E. and Tegegne, F. 2015. Soil erosion and conservation in Ethiopia: a review. *Progress in Physical Geography*, *39*(6), pp.750-774.

Haregeweyn, N., Tsunekawa, A., Poesen, J., Tsubo, M., Meshesha, D.T., Fenta, A.A., Nyssen, J. and Adgo, E. 2017. Comprehensive assessment of soil erosion risk for better land use planning in river basins: Case study of the Upper Blue Nile River. *Science of the Total Environment*, *574*, pp.95-108.

Harris, T. and Consulting, T.H. 2014. *Africa agriculture status report 2014: Climate change and smallholder agriculture in Sub-Saharan Africa* (No. BOOK). Alliance for a Green Revolution in Africa (AGRA).

Hartemink, A.E. 2010. Land use change in the tropics and its effect on soil fertility. In *Proceedings 19th World Congress of Soil Science, Brisbane, Australia, 01-06 August, 2010,* pp. 55-58.

Hassen, E.E. and Assen, M. 2018. Land use/cover dynamics and its drivers in Gelda catchment, Lake Tana watershed, Ethiopia. *Environmental Systems Research*, *6*(1), pp.1-13.

Haylemariyam, M. B. 2018. Detection of Land Surface Temperature in Relation to Land Use Land Cover Change: Dire Dawa City, Ethiopia. *J Remote Sens GIS*, 7: 245.

He, G., Zhao, Y., Wang, J., Wang, Q. and Zhu, Y. 2018. Impact of large-scale vegetation restoration project on summer land surface temperature on the Loess Plateau, China. *Journal of Arid Land*, *10*, pp.892-904.

Hegazy, I.R. and Kaloop, M.R. 2015. Monitoring urban growth and land use change detection with GIS. *International Journal of Sustainable Built Environment*, pp.177-124.

Helldén, U. 1987. An assessment of woody biomass, community forests, land use and soil erosion in Ethiopia. A feasibility study on the use of remote sensing and GIS [geographical information system]-analysis for planning purposes in developing countries. Lund University Press, Lund.

Henao, J. and Baanante, C. 2006. Agricultural production and soil nutrient mining in Africa: Implications for resource conservation and policy development.

Hengl, T., Heuvelink, G.B., Kempen, B., Leenaars, J.G., Walsh, M.G., Shepherd, K.D., Sila, A., MacMillan, R.A., Mendes de Jesus, J., Tamene, L. and Tondoh, J.E. 2015. Mapping soil properties of Africa at 250 m resolution: Random forests significantly improve current predictions. *PloS one*, *10*(6), p.e0125814.

Herweg, K. and Stillhardt, B. 1999. The variability of soil erosion in the Highlands of Ethiopia and Eritrea. *Average and Extreme Erosion Patterns, University of Berne, Berne*.

Hua, A.K. and Ping, O.W. 2018. The influence of land-use/land-cover changes on land surface temperature: a case study of Kuala Lumpur metropolitan city. *European Journal of Remote Sensing*, *51*(1), pp.1049-1069.

Huang, W., Zong, M., Fan, Z., Feng, Y., Li, S., Duan, C. and Li, H. 2021. Determining the impacts of deforestation and corn cultivation on soil quality in tropical acidic red soils using a soil quality index. *Ecological Indicators*, *125*, p.107580.

Hurni, H. 1985. Erosion-productivity-conservation systems in Ethiopia.

Hurni, H. 1986. Management Plan for Simen Mountains National Park and Surrounding Rural Area.

Hurni, H. 1993. Land degradation, famine, and land resource scenarios in Ethiopia. *World soil erosion and conservation.*, pp.27-61.

Hurni, H. and Meyer, K. 2002. *A world soils agenda: discussing international actions for the sustainable use of soils*. Geographica Bernensia, Bern.

Hurni, H., Giger, M., Liniger, H., Studer, R.M., Messerli, P., Portner, B., Schwilch, G., Wolfgramm, B. and Breu, T. 2015. Soils, agriculture and food security: the interplay between ecosystem functioning and human well-being. *Current Opinion in Environmental Sustainability*, *15*, pp.25-34.

Igun, E. and Williams, M. 2018. Impact of urban land cover change on land surface temperature. *Global Journal of Environmental Science and Management*, *4*(1), pp.47-58.

Imran, H.M., Hossain, A., Islam, A.S., Rahman, A., Bhuiyan, M.A.E., Paul, S. and Alam, A. 2021. Impact of Land Cover Changes on Land Surface Temperature and Human Thermal Comfort in Dhaka City of Bangladesh. *Earth Systems and Environment*, pp.1-27.

IUSS Working Group, W.R.B. 2006. World reference base for soil resources. *World Soil Resources Report*, *103*.

Jain, M., Dimri, A.P. and Niyogi, D. 2017. Land-air interactions over urban-rural transects using satellite observations: analysis over Delhi, India from 1991–2016. *Remote Sensing*, *9*(12), p.1283.

Jeevalakshmi, D., Narayana Reddy, S. and Manikiam, B. 2017. Land Surface Temperature Retrieval from LANDSAT data using Emissivity Estimation. *International Journal of Applied Engineering Research*, 12(20), pp. 9679-9687.

Julio, H. and Carlos, B. 2006. Agricultural production and soil nutrient mining in Africa: implication for resource conservation and policy development. *International Centre for Soil Fertility and Agriculture Development. Alabama, USA*.

Julien PY, Frenette M (1998) Physical processes governing reservoir sedimentation. In: Conference on reservoir sedimentation (pp. 121–142). Fort Collins, Colorado.

Kafy, A.A., Rahman, M.S., Hasan, M.M. and Islam, M. 2020. Modelling future land use land cover changes and their impacts on land surface temperatures in Rajshahi, Babgladish. Remote Sens. Appli.: Soc.Environ.18, p.100314.

Kandel, D.D., Western, A.W., Grayson, R.B. and Turral, H.N., 2004. Process parameterization and temporal scaling in surface runoff and erosion modelling. *Hydrological Processes*, *18*(8), pp.1423-1446.

Karaburun, A. 2010. Estimation of C factor for soil erosion modeling using NDVI in Buyukcekmece watershed. *Ozean Journal of applied sciences*, *3*(1), pp.77-85.

Karaca, S., Dengiz, O., Turan, İ.D., Özkan, B., Dedeoğlu, M., Gülser, F., Sargin, B., Demirkaya, S. and Ay, A. 2021. An assessment of pasture soils quality based on multi-indicator weighting approaches in semi-arid ecosystem. *Ecological Indicators*, *121*, p.107001.

Karlen, D.L., Tomer, M.D., Neppel, J. and Cambardella, C.A. 2008. A preliminary watershed scale soil quality assessment in north central Iowa, USA. *Soil and Tillage Research*, *99*(2), pp.291-299.

Keating, B. 1997. Monitoring cane at the mill to improve nitrogen management on the farm, a pilot study: final report CSC21A.

Kebede, Y.S., Endalamaw, N.T., Sinshaw, B.G. and Atinkut, H.B. 2021. Modeling soil erosion using RUSLE and GIS at watershed level in the upper beles, Ethiopia. *Environmental Challenges*, *2*, p.100009.

Kidane, M., Bezie, A., Kesete, N. and Tolessa, T. 2019. The impact of land use and land cover (LULC) dynamics on soil erosion and sediment yield in Ethiopia. *Heliyon*, *5*(12), p.e02981.

Kikon, N., Singh, P., Singh, S.K. and Vyas, A. 2016. Assessment of urban heat islands (UHI) of Noida City, India using multi-temporal satellite data. *Sustainable Cities and Society*, *22*, pp.19-28.

Kimuku, C.W. and Ngigi, M.M. 2017. Study of urban heat island trends to aid in urban planning in Nakuru County-Kenya.

Kiros, G., Mitiku, H. and Girmay, G. 2014. "Assessing the input and output flows and nutrients balance analysis at catchment level in Northern Ethiopia." *Journal of Soil Science and Environmental Management,* 5 (1), pp. 1-12.

Kouli, M., Soupios, P. and Vallianatos, F. 2009. Soil erosion prediction using the revised universal soil loss equation (RUSLE) in a GIS framework, Chania, Northwestern Crete, Greece. *Environmental geology*, *57*(3), pp.483-497.

Kumar, S. and Mishra, A. 2015. Critical erosion area identification based on hydrological response unit level for effective sedimentation control in a river basin. *Water Resources Management*, *29*(6), pp.1749-1765.

Landon, J.R. 2014. *Booker tropical soil manual: a handbook for soil survey and agricultural land evaluation in the tropics and subtropics*. Routledge, p. 228.

Larson, W.E. and Pierce, F.J. 1994. The dynamics of soil quality as a measure of sustainable management. *Defining soil quality for a sustainable environment*, *35*, pp.37-51.

Lefroy, R. D. and Wijnhoud, J. 2001. Nutrient balance studies: General use and perspectives for SE Asia. Paper presented at the International Workshop on Nutrient Balances for Sustainable Agricultural Production and Natural Resource Management in Southeast Asia, 20-22 February 2001, Bangkok, Thailand.

Lepcha, N.T. and Devi, N.B. 2020. Effect of land use, season, and soil depth on soil microbial biomass carbon of Eastern Himalayas. *Ecological processes*, *9*(1), pp.1-14.

Lesschen, J.P., Stoorvogel, J.J., Smaling, E.M.A., Heuvelink, G.B.M. and Veldkamp, A. 2007. A spatially explicit methodology to quantify soil nutrient balances and their uncertainties at the national level. *Nutrient Cycling in Agroecosystems*, *78*(2), pp.111-131.

Lewoyehu, M., Alemu, Z. and Adgo, E. 2020. The effects of land management on soil fertility and nutrient balance in Kecha and Laguna micro watersheds, Amhara Region, Northwestern, Ethiopia. *Cogent Food & Agriculture*, *6*(1), p.1853996.

Li, T., Wang, G., Xue, H. and Wang, K. 2009. Soil erosion and sediment transport in the gullied Loess Plateau: scale effects and their mechanisms. *Science in China Series E: Technological Sciences*, *52*(5), pp.1283-1292.

Li, K., Feng, M., Biswas, A., Su, H., Niu, Y. and Cao, J. 2020. Driving factors and future prediction of land use and cover change based on satellite remote sensing data by the LCM model: a case study from Gansu province, China. *Sensors*, *20*(10), p.2757.

Li, X., Li, H., Yang, L. and Ren, Y. 2018. Assessment of soil quality of crop lands in the corn belt of Northwest China. S*ustainability*, 10, 248.

Manaouch, M., Sadiki, M. and Fenjiro, I. 2021. Integrating GIS-based FAHP and WaTEM/SEDEM for identifying potential RWH areas in semi-arid areas. *Geocarto International*, pp.1-24.

Mancino, G., Nole, A., Ripullone, F. and Ferrara, A. 2014. Landsat TM imagery and NDVI differencing to detect vegetation change: Assessing natural forest expansion in Basilicata, southern Italy. Iforest-Biogeosci. For., 7(2), 76–85.

Markose, V.J. and Jayappa, K.S. 2016. Soil loss estimation and prioritization of sub-watersheds of Kali River basin, Karnataka, India, using RUSLE and GIS. *Environmental monitoring and assessment*, *188*(4), pp.1-16.

Marques, S. M. 2021. Modelling Sediment Retention Services and Soil Erosion Changes in Portugal: A Spatio-Temporal Approach. ISPRS, *Int. J. Geo-Inf.*, 10, 262.

Marttila, H. and Kløve, B. 2010. Dynamics of erosion and suspended sediment transport from drained peat land forestry. *Journal of hydrology*, *388*(3-4), pp.414-425.

Masto, R.E., Sheik, S., Nehru, G., Selvi, V.A., George, J. and Ram, L.C. 2015. Assessment of environmental soil quality around Sonepur Bazari mine of Raniganj coalfield, India. *Solid Earth*, *6*(3), pp.811-821.

Maurya, S., Abraham, J.S., Somasundaram, S., Toteja, R., Gupta, R. and Makhija, S., 2020. Indicators for assessment of soil quality: a mini-review. *Environmental Monitoring and Assessment*, *192*, pp.1-22.

McCool, D.K., Foster, G.R., Mutchler, C.K. and Meyer, L.D. 1989. Revised slope length factor in the Universal Soil Loss Equation. Trans, ASAE 30: 1571-1576.

Mekonnen, M., Keesstra, S.D., Baartman, J.E., Stroosnijder, L. and Maroulis, J. 2017. Reducing sediment connectivity through man-made and natural sediment sinks in the Minizr catchment, Northwest Ethiopia. *Land degradation & development*, *28*(2), pp.708-717.

Mekonnen, Y.A., Mengistu, T.D., Asitatikie, A.N. and Kumilachew, Y.W. 2022. Evaluation of reservoir sedimentation using bathymetry survey: a case study on Adebra night storage reservoir, Ethiopia. *Applied Water Science*, *12*(12), p.269.

Meng, X., Cheng, J., Zhao, S., Liu, S. and Yao, Y. 2019. Estimating land surface temperature from Landsat-8 data using the NOAA JPSS enterprise algorithm. *Remote Sensing*, *11*(2), p.155.

Merritt, W.S., Letcher, R.A. and Jakeman, A.J. 2003. A review of erosion and sediment transport models. *Environmental modelling & software*, *18*(8-9), pp.761-799.

Mesfin, S., Gebresamuel, G., Zenebe, A. and Haile, M. 2021. Nutrient balances in smallholder farms in northern Ethiopia. *Soil Use and Management*, 37(3), 468-478.

Meshesha, D.T., Tsunekawa, A., Tsubo, M., Ali, S.A. and Haregeweyn, N. 2014. Land-use change and its socio-environmental impact in Eastern Ethiopia's highland. *Regional environmental change*, *14*, pp.757-768.

Ministry of Water Resources (MoWR). 2011. Abay Basin Integrated Development Master Plan main report, Addis Ababa, Ethiopia.

Minta, M., Kibret, K., Thorne, P., Nigussie, T. and Nigatu, L. 2018. Land use and land cover dynamics in Dendi-Jeldu hilly-mountainous areas in the central Ethiopian highlands. *Geoderma*, *314*, pp.27-36.

Mohammed, S., Alsafadi, K., Talukdar, S., Kiwan, S., Hennawi, S., Alshihabi, O., Sharaf, M. and Harsanyie, E. 2020. Estimation of soil erosion risk in southern part of Syria by using RUSLE integrating geo informatics approach. *Remote Sensing Applications: Society and Environment*, *20*, p.100375.

Moisa, M.B., Negash, D.A., Merga, B.B. and Gemeda, D.O. 2021. Impact of land-use and land-cover change on soil erosion using the RUSLE model and the geographic information system: a case of Temeji watershed, Western Ethiopia. *Journal of Water and Climate Change*, *12*(7), pp.3404-3420.

Molla, E., Getnet, K. and Mekonnen, M. 2022. Land use change and its effect on selected soil properties in the northwest highlands of Ethiopia. *Heliyon*, *8*(8), p.e10157.

Mondal, A., Kundu, S., Chandniha, S.K., Shukla, R. and Mishra, P.K. 2012. Comparison of support vector machine and maximum likelihood classification technique using satellite imagery. *International Journal of Remote Sensing and GIS*, *1*(2), pp.116-123.

Monserud, R. A. and Rik, L. 1992. "Comparing global vegetation maps with the Kappa statistic." *Ecological modelling* 62 (4), 275-293.

Moore, I.D. and Burch, G.J. 1986a. Physical basis of the length-slope factor in the universal soil loss equation. *Soil Science Society of America Journal*, *50*(5), pp.1294-1298.

Morgan, R.P.C. 1996. Verificaton of the European Soil Erosion Model (EOROSEM) for varying slope and vegetation conditions. *Advances in hillslope processes: volume 1.*, pp.657-668.

Mujabar, P.S., 2019. Spatial-temporal variation of land surface temperature of Jubail Industrial City, Saudi Arabia due to seasonal effect by using Thermal Infrared Remote Sensor (TIRS) satellite data. *Journal of African Earth Sciences*, *155*, pp.54-63.

Mukherjee, A. and Lal, R. 2014. Comparison of soil quality index using three methods. *PloS one*, *9*(8), p.e105981.

Mulat, Y., Kibret, K., Bedadi, B. and Mohammed, M. 2021. Soil quality evaluation under different land use types in Kersa sub-watershed, eastern Ethiopia. *Environmental Systems Research*, *10*(1), pp.1-11.

Mulualem, T., Adgo, E., Meshesha, D.T., Tsunekawa, A., Haregeweyn, N., Tsubo, M., Ebabu, K., Kebede, B., Berihun, M.L., Walie, M. and Mekuriaw, S. 2021. Exploring the variability of soil nutrient outflows as influenced by land use and management practices in contrasting agroecological environments. *Science of the Total Environment*, *786*, p.147450.

Munyaneza, O. 2015. Estimation of the Climate Variability Impact on Water Resources in the Nyabugogo Swamp in Rwanda.

Mutua, B.M., Klik, A. and Loiskandl, W. 2006. Modelling soil erosion and sediment yield at a catchment scale: the case of Masinga catchment, Kenya. *Land degradation & development*, *17*(5), pp.557-570.

Nandwa, S.M. and Bekunda, M.A. 1998. Research on nutrient flows and balances in East and Southern Africa: state-of-the-art. *Agriculture, ecosystems & environment*, *71*(1-3), pp.5-18.

Narendra, B.H., Siringoringo, H.H. and Siregar, C.A. 2017. GIS Based Flood Hazard and Vulnerability Mapping: A Case Study of Tidal and River Floods in Downstream of Ciasem Watershed, Subang-West Java. *Indonesian Journal of Forestry Research*, *4*(1), pp.37-48.

Nath, B., Niu, Z. and Singh, R.P. 2018. Land Use and Land Cover changes, and environment and risk evaluation of Dujiangyan city (SW China) using remote sensing and GIS techniques. *Sustainability*, *10*(12), p.4631.

Neglo, K.A.W., Gebrekidan, T. and Lyu, K. 2021. Determinants of participation in non-farm activities and its effect on household income: An empirical study in Ethiopia. *Journal of Development and Agricultural Economics*, *13*(1), pp.72-92.

Ngo-Mbogba, M., Yemefack, M. and Nyeck, B. 2015. Assessing soil quality under different land cover types within shifting agriculture in South Cameroon. *Soil and tillage research*, *150*, pp.124-131.

Nguemezi, C., Tematio, P., Yemefack, M., Tsozue, D. and Silatsa, T.B.F. 2020. Soil quality and soil fertility status in major soil groups at the Tombel area, South-West Cameroon. *Heliyon*, *6*(2), p.e03432.

Nkonya, E. Place, F. Kato, E. Mwanjololo, M. 2015. Climate risk management through sustainable land management in Sub-Saharan Africa. In: Lal R, Singh BR, Mwaseba DL, Kraybill D, Hansen DO, Eik LO (eds) Sustainable intensification to advance food security and enhance climate resilience in Africa. Springer, Cham, pp 75–111.

Norman, J.M. and Becker, F. 1995.Terminology in thermal infrared remote sensing of natural surfaces. *Agricultural and Forest Meteorology*, *77*(3-4), pp.153-166.

Nyssen, J., Clymans, W., Poesen, J., Vandecasteele, I., De Baets, S., Haregeweyn, N., Naudts, J., Hadera, A., Moeyersons, J., Haile, M. and Deckers, J. 2009. How soil conservation affects the catchment sediment budget–a comprehensive study in the north Ethiopian highlands. *Earth Surface Processes and Landforms*, *34*(9), pp.1216-1233.

Obalum, S.E., Buri, M.M., Nwite, J.C., Watanabe, Y., Igwe, C.A. and Wakatsuki, T. 2012. Soil degradation-induced decline in productivity of sub-Saharan African soils: the prospects of looking downwards the lowlands with the Sawah ecotechnology. *Applied and Environmental Soil Science, volume 12.*

Olsen, S.R., Watanabe, F.S., Cosper, H.R., Larson, W.E. and Nelson, L.B. 1954. Residual phosphorus availability in long-time rotations on calcareous soils. *Soil Science*, *78*(2), pp.141-152.

Onduru, D.D., De Jager, A., Muchena, F.N., Gachimbi, L. and Gachini, G.N. 2007. Socio-economic factors, soil fertility management and cropping practices in mixed farming systems of sub-Saharan Africa: A study in Kiambu, central highlands of Kenya. *International journal of agricultural research*, *2*(5), pp.426-439.

Onduru, D.D. and Du Preez, C.C. 2007. Spatial and temporal aspects of agricultural sustainability in the semi-arid tropics: a case study in Mbeere district, Eastern Kenya. *Tropical Science*, *47*(3), pp.134-148.

Opeyemi, O.A., Abidemi, F.H. and Victor, O.K. 2019. Assessing the impact of soil erosion on residential areas of Efon-Alaaye Ekiti, Ekiti-State, Nigeria. *Int. J. Environ. Plan. Manag*, *5*, pp.23-31.

Ota, H.O., Aja, D., Okolo, C.C., Oranu, C.O. and Nwite, J.N. 2018. Influence of tree plantation Gmelina Arborea and Gliricidia sepium on soil physico-chemical properties in Abakaliki, Southeast, Nigeria. *Acta Chemica Malaysia (ACMY)*, *2*(2), pp.23-28.

Padbhushan, R., Kumar, U., Sharma, S., Rana, D.S., Kumar, R., Kohli, A., Kumari, P., Parmar, B., Kaviraj, M., Sinha, A.K. and Annapurna, K. 2022. Impact of land-use changes on soil properties and carbon pools in India: A meta-analysis. *Frontiers in Environmental Science*, *9*, p.794866.

Pal, S. and Ziaul, S.K. 2017. Detection of land use and land cover change and land surface temperature in English Bazar urban centre. *The Egyptian Journal of Remote Sensing and Space Science*, *20*(1), pp.125-145.

Panagos, P., Meusburger, K., Ballabio, C., Borrelli, P. and Alewell, C. 2014. Soil erodibility in Europe: A high-resolution dataset based on LUCAS. *Science of the total environment*, *479*, pp.189-200.

Patel, S.K., Verma, P. and Singh, G.S. 2019. Agricultural growth and land use land cover change in peri-urban India. *Environmental monitoring and assessment*, *191*(9), pp.1-17.

Peng, X., Zhu, Q.H., Xie, Z.B., Darboux, F. and Holden, N.M. 2016. The impact of manure, straw and biochar amendments on aggregation and erosion in a hillslope Ultisol. *Catena*, *138*, pp.30-37.

Pham, T.G., Nguyen, H.T. and Kappas, M. 2018. Assessment of soil quality indicators under different agricultural land uses and topographic aspects in Central Vietnam. *International Soil and Water Conservation Research*, *6*(4), pp.280-288.

Pieri, C.J. 1992. Farming and Physical Degradation of the Soils. *In Fertility of Soils*, pp. 127-131. Springer, Berlin, Heidelberg.

Rahman, G.K.M., Rahman, M.M., Alam, M.S., Kamal, M.Z., Mashuk, H.A., Datta, R. and Meena, R.S. 2020. Biochar and organic amendments for sustainable soil carbon and soil health. *In Carbon and nitrogen cycling in soil*, pp. 45-85. Springer, Singapore.

Rajani, A. and Varadarajan, S. 2021. Estimation and Validation of Land Surface Temperature by using Remote Sensing and GIS for Chittoor District, Andhra Pradesh. *Turkish Journal of Computer and Mathematics Education*, *12*(5), pp.607-617.

Renard, K.G. 1997. Predicting soil erosion by water*: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE)*. United States Government Printing.

Renard, K.G., Yoder, D.C., Lightle, D.T. and Dabney, S.M. 2011. Univeral Soil Loss Equation and Revised Universal Soil Loss Equation. *Handbook of Erosion Modelling, edited by: Morgan, R. PC and Nearing, MA*, pp.137-167.

Rientjes, T.H.M., Haile, A.T., Kebede, E., Mannaerts, C.M.M., Habib, E. and Steenhuis, T.S. 2011. Changes in land cover, rainfall and stream flow in Upper Gilgel Abbay catchment, Blue Nile basin–Ethiopia. *Hydrology and Earth System Sciences*, *15*(6), pp.1979-1989.

Rongali, G., Keshari, A.K., Gosain, A.K. and Khosa, R. 2018. A Mono-Window Algorithm for Land Surface Temperature Estimation from Landsat 8 Thermal Infrared Sensor Data: A Case Study of the Beas River Basin, India, *Pertanika J. Sci. & Technol*. 26 (2): 829 – 840.

Roy, R.N., Misra, R.V., Lesschen, J.P and Smaling, EM.A. 2003. Assessment of soil nutrient balance. Approaches and methodologies. FAO Fertilizer and plant nutrition bulletins 14. FAO, Rome.

Rozos, D., Skilodimou, H.D., Loupasakis, C. and Bathrellos, G.D. 2013. Application of the revised universal soil loss equation model on landslide prevention. An example from N. Euboea (Evia) Island, Greece. *Environmental Earth Sciences*, *70*(7), pp.3255-3266.

Sahana, M., Ahmed, R. and Sajjad, H. 2016. Analyzing land surface temperature distribution in response to land use/land cover change using split window algorithm and spectral radiance model in Sundarban Biosphere Reserve, India. *Modeling Earth Systems and Environment*, *2*, pp.1-11.

Sahlemedhin, S. and Taye, B. 2000. Procedures for soil and plant analysis. *Technical paper*, *74*, p.110.

Sanderman, J., Hengl, T. and Fiske, G.J. 2017. Soil carbon debt of 12,000 years of human land use. *Proceedings of the National Academy of Sciences*, *114*(36), pp.9575-9580.

Schuch, F., Marpu, P., Masri, D. and Afshari, A., 2017. Estimation of urban air temperature from a rural station using remotely sensed thermal infrared data. *Energy Procedia*, *143*, pp.519-525.

SCRP (Soil Conservation Research Project). 1996. Soil erosion hazard assessment for land evaluation. Research report, SCRP, Addis Ababa, Ethiopia.

Seely, B., Welham, C. and Blanco, J.A. 2010. Towards the application of soil organic matter as an indicator of forest ecosystem productivity: Deriving thresholds, developing monitoring systems, and evaluating practices. *Ecological Indicators*, *10*(5), pp.999-1008.

Seifu, W., Elias, E. and Gebresamuel, G. 2020. The effects of land use and landscape position on soil physicochemical properties in a semiarid watershed, northern Ethiopia. *Applied and Environmental Soil Science*, *2020*, pp.1-20.

Selassie, Y.G. and Belay, Y. 2013. Costs of nutrient losses in priceless soils eroded from the highlands of Northwestern Ethiopia. *Journal of Agricultural Science*, *5*(7), p.227.

Setegn, S.G., Srinivasan, R., Dargahi, B. and Melesse, A.M. 2009. Spatial delineation of soil erosion vulnerability in the Lake Tana Basin, Ethiopia. *Hydrological Processes: An International Journal*, *23*(26), pp.3738-3750.

Sharda, V.N. and Ojasvi, P.R. 2016. A revised soil erosion budget for India: role of reservoir sedimentation and land-use protection measures. *Earth Surface Processes and Landforms*, *41*(14), pp.2007-2023.

Sharp, R. Tallis, H.T. Ricketts, T. Guerry, A.D. Wood, S.A. Chaplin-Kramer, R. Nelson, E. Ennaanay, D. Wolny, S. Olwero, N. and Douglass, J. 2018. InVEST User's Guide. The Natural Capital Project, Stanford University, University of Minnesota, the Nature Conservancy, and World Wildlife Fund.

Shastri, S., Singh, P., Verma, P., Rai, P.K. and Singh, A.P. 2020. Land cover change dynamics and their impacts on thermal environment of Dadri block, Gautam budh Nagar, India. *Journal of Landscape Ecology*, *13*(2), pp.1-13.

Sheikh, M.A., Kumar, M., Bussman, R.W. and Todaria, N.P. 2011. Forest carbon stocks and fluxes in physiographic zones of India. *Carbon balance and management*, *6*(1), pp.1-10.

Sheldrick, W.F. and Lingard, J. 2004. The use of nutrient audits to determine nutrient balances in Africa. *Food Policy*, *29*(1), pp.61-98.

Sheldrick, W.F., Syers, J.K. and Lingard, J. 2003. Soil nutrient audits for China to estimate nutrient balances and output/input relationships. *Agriculture, ecosystems & environment*, *94*(3), pp.341-354.

Shen, P., Zhang, L.M., Chen, H.X. and Gao, L. 2017. Role of vegetation restoration in mitigating hill slope erosion and debris flows. *Engineering Geology*, *216*, pp.122-133.

Shibabaw, A. and Alemeyehu, M. 2015. The Contribution of Some Soil and Crop Management Practice on Soil Organic Carbon Reserves.

Shukla, M.K., Lal, R. and Ebinger, M. 2006. Determining soil quality indicators by factor analysis. *Soil and Tillage Research, 87*(2), pp.194-204.

Sileshi, M., Kadigi, R., Mutabazi, K. and Sieber, S. 2019. Determinants for adoption of physical soil and water conservation measures by smallholder farmers in Ethiopia. *International soil and water conservation research*, *7*(4), pp.354-361.

Simane, B., Zaitchik, B.F. and Ozdogan, M. 2013. Agroecosystem analysis of the Choke Mountain watersheds, Ethiopia. *Sustainability*, *5*(2), pp.592-616.

Six, J. and Paustian, K. 2014. Aggregate-associated soil organic matter as an ecosystem property and a measurement tool. *Soil Biology and Biochemistry*, *68*, pp.A4-A9.

Smaling EMA, Lesschen JP, van Beek CL, De Jager A, Stoorvogel JJ, Batjes NH, Fresco LO (2013) Where do we stand 20 years after the assessment of soil nutrient balances in sub-Saharan Africa? In: Lal R, Steart BA (eds) World soil resources and food security. CRC press, Taylor & Francis Group, Boca Raton.

Smaling, E.M.A., Stoorvogel, J.J. and Windmeijer, P.N., 1993. Calculating soil nutrient balances in Africa at different scales: II. District scale. *Fertilizer Research*, *35*, pp.237-250.

Sobrino, J.A. and Romaguera, M., 2004.Land surface temperature retrieval from MSG1-SEVIRI data. *Remote Sensing of Environment*, *92*(2), pp.247-254.

Song, Y., Liu, L., Yan, P. and Cao, T. 2005. A review of soil erodibility in water and wind erosion research. *Journal of Geographical Sciences*, *15*(2), pp.167-176.

Sonneveld, B.G., Keyzer, M.A. and Stroosnijder, L. 2011. Evaluating quantitative and qualitative models: an application for nationwide water erosion assessment in Ethiopia. *Environmental Modelling & Software*, *26*(10), pp.1161-1170.

Stoorvogel, J.J. and Smaling, E.M.A. 1990. Assessment of soil nutrient depletion in Sub-Saharan Africa: *1983-2000. Vol. 1: Main report* (No. 28). SC-DLO.

Tadesse, A and Abebe, M. 2014. GIS based soil loss estimation using RUSLE Model: the case of Jabi Tehinan Woreda, ANRS. Ethiopia. *Natural Resource*, 5:616–626.

Tadesse, L., Suryabhagavan, K.V., Sridhar. G. and Legesse, G. 2017. Land use and land cover changes and Soil erosion in Yezat Watershed, North Western Ethiopia. *International soil and water conservation research*, *5*(2), pp.85-94.

Tahir, M., Imam, E. and Hussain, T. 2013. Evaluation of land use/land cover changes in Mekelle City, Ethiopia using Remote Sensing and GIS. *Computational Ecology and Software*, *3*(1), p.9.

Tamene, L., Adimassu, Z., Aynekulu, E. and Yaekob, T. 2017. Estimating landscape susceptibility to soil erosion using a GIS-based approach in Northern Ethiopia. *International Soil and Water Conservation Research*, *5*(3), pp.221-230.

Tamene, L. and Le, Q.B. 2015. Estimating soil erosion in sub-Saharan Africa based on landscape similarity mapping and using the revised universal soil loss equation (RUSLE). *Nutrient Cycling in Agroecosystems*, *102*, pp.17-31.

Tamene, L. and Vlek, P.L. 2008. Soil erosion studies in Northern Ethiopia. *Center for Development Research*, University of Bonn, Bonn.

Tamene, L., Park, S.J., Dikau, R. and Vlek, P.L.G. 2006. Analysis of factors determining sediment yield variability in the highlands of northern Ethiopia. *Geomorphology*, *76*(1-2), pp.76-91.

Tan, Z.X., Lal, R. and Wiebe, K.D.2005. Global soil nutrient depletion and yield reduction. *Journal of sustainable agriculture*, *26*(1), pp.123-146.

Taye, G., Poesen, J., Van Wesemael, B., Goosse, T., Teka, D., Deckers, J., Hallet, V., Haregeweyn, N., Nyssen, J. and Maetens, W. 2013. Effects of land use, slope gradient, and soil and water conservation techniques, on runoff and soil loss in a semi-arid environment. Journal of Physical Geography 34 (3), 236–259.

Teferi, E., Bewket, W. and Simane, B. 2016. Effects of land use and land cover on selected soil quality indicators in the headwater area of the Blue Nile basin of Ethiopia. *Environmental monitoring and assessment*, *188*(2), pp.1-12.

Teferi, E., Bewket, W., Uhlenbrook, S. and Wenninger, J. 2013. Understanding recent land use and land cover dynamics in the source region of the Upper Blue Nile, Ethiopia: Spatially explicit statistical modeling of systematic transitions. *Agriculture, ecosystems & environment*, *165*, pp.98-117.

Tegene, B. 2002. Land-cover/land-use changes in the derekolli catchment of the South Welo Zone of Amhara Region, Ethiopia. *Eastern Africa Social Science Research Review*, *18*(1), pp.1-20.

Tekle, K. and Hedlund, L. 2000. Land cover changes between 1958 and 1986 in Kalu District, southern Wello, Ethiopia. *Mountain research and development*, *20*(1), pp.42-51.

Tellen, V.A. and Yerima, B.P. 2018. Effects of land use change on soil physicochemical properties in selected areas in the North West region of Cameroon. *Environmental systems research*, *7*(1), pp.1-29.

Temesgen, G., Amare, B. and Silassie, H.G. 2014a. Land degradation in Ethiopia: causes, impacts and rehabilitation techniques. *Journal of environment and earth science*, *4*(9), pp.98-104.

Tiruneh, G. and Ayalew, M. 2015. Soil loss estimation using geographic information system in enfraz watershed for soil conservation planning in highlands of Ethiopia. *International Journal of Agricultural Research, Innovation and Technology*, *5*(2), pp.21-30.

Traore, M., Lee, M.S., Rasul, A. and Balew, A. 2021. Assessment of land use/land cover changes and their impacts on land surface temperature in Bangui (the capital of Central African Republic). *Environmental Challenges*, *4*, p.100114.

Tsegaye, B. 2019. Effect of land use and land cover changes on soil erosion in Ethiopia. *International Journal of Agricultural Science and Food Technology*, *5*(1), pp.026-034.

Tsegaye, D., Moe, S.R., Vedeld, P. and Aynekulu, E. 2010. Land use/cover dynamics in Northern Afar rangelands, Ethiopia. *Agriculture, ecosystems & environment*, *139*(1-2), pp.174-180.

Tufa, M., Melese, A. and Wondwosen, T. 2019. Effects of land use types on selected soil physical and chemical properties: The case of Kuyu District, Ethiopia. *Eurasian Journal of Soil Science*, 8 (2) 94-109.

Tully, K., Sullivan, C., Weil, R. and Sanchez, P. 2015. The state of soil degradation in Sub-Saharan Africa: Baselines, trajectories, and solutions. *Sustainability*, *7*(6), pp.6523-6552.

Twisa, S. and Buchroithner, M.F. 2019. Land-use and land-cover (LULC) change detection in Wami River Basin, Tanzania. *Land*, *8*(9), p.136.

Uddin, J., Mohiuddin, A.S.M. and Hassan, M. 2019. Organic carbon storage in the tropical peat soils and its impact on climate change. *Am J Clim Change*, *8*, pp.94-109.

USDA NRCS .2020. "Carbon to nitrogen ratios in cropping systems," *USDA Natural Resources Conservation Service, Soils*, vol. 47.

Van Beek, C. L., Elias, E., Yihenew, G. S., Heesmans, H., Tsegaye, A., Feyisa, H., Tolla, M., Melmuye, M., Gebremeskel, Y. and Mengist, S. 2016. Soil nutrient balances under diverse agroecological settings in Ethiopia, *Nutr. Cycl. Agroecosyst*, 106, pp.257–274.

Van Breemen, N.V., Boyer, E.W., Goodale, C.L., Jaworski, N.A., Paustian, K., Seitzinger, S.P., Lajtha, K., Mayer, B., Van Dam, D., Howarth, R.W. and Nadelhoffer, K.J. 2002. Where did all the nitrogen go? Fate of nitrogen inputs to large watersheds in the northeastern USA. *Biogeochemistry*, *57*, pp.267-293.

Van de Griend, A.A. and Owe, M. 1993. On the relationship between thermal emissivity and the normalized difference vegetation index for natural surfaces. *International Journal of remote sensing*, *14*(6), pp.1119-1131.

Van Reeuwijk L. 2006. *Procedures for soil analysis* (6th edn), Wageningen: International Soil Reference and Information Centre.

Vasu, D., Singh, S.K., Ray, S.K., Duraisami, V.P., Tiwary, P., Chandran, P., Nimkar, A.M. and Anantwar, S.G. 2016. Soil quality index (SQI) as a tool to evaluate crop productivity in semi-arid Deccan plateau, India. *Geoderma*, *282*, pp.70-79.

Vitousek, P.M., Naylor, R., Crews, T., David, M.B., Drinkwater, L.E., Holland, E., Johnes, P.J., Katzenberger0, J., Martinelli, L.A., Matson, P.A. and Nziguheba, G. 2009. Nutrient imbalances in agricultural development. *Science*, *324*(5934), pp.1519-1520.

Vlek, P.L., Le, Q.B. and Tamene, L. 2008. African land degradation in a global atmospheric change. In *Sustainable Development in Drylands–Meeting the Challenge of Global Climate Change*, November 9, (Vol. 7, p. 104).

Walkley, A. and Black, I.A. 1934. An examination of the Degtjareff method for determining soil organic matter, and a proposed modification of the chromic acid titration method. *Soil science*, *37*(1), pp.29-38.

Wang, X., Zhao, X., Zhang, Z., Yi, L., Zuo, L., Wen, Q., Liu, F., Xu, J., Hu, S. and Liu, B. 2016. Assessment of soil erosion change and its relationships with land use/cover change in China from the end of the 1980s to 2010. *Catena*, *137*, pp.256-268.

Wang, F., Qin, Z., Song, C., Tu, L., Karnieli, A. and Zhao, S. 2015. An improved mono-window algorithm for land surface temperature retrieval from Landsat 8 thermal infrared sensor data. *Remote sensing,* 7**,** 4268-4289.

Wang, J., Fu, B., Qiu, Y. and Chen, L. 2001. Soil nutrients in relation to land use and landscape position in the semi-arid small catchment on the loess plateau in China. *Journal of arid environments*, *48*(4), pp.537-550.

Warra, H.H., Ahmed, M.A. and Nicolau, M.D. 2015. Impact of land cover changes and topography on soil quality in the Kasso catchment, Bale Mountains of southeastern Ethiopia. *Singapore Journal of Tropical Geography*, *36*(3), pp.357-375.

Wasige, E.J. 2013. A spatially explicit approach to determine hydrology, erosion, and nutrient dynamics in an upstream catchment of lake victoria basin. *University of Twente Faculty of Geo-Information and Earth Observation (ITC), Enschede, ITC Dissertation, 239.*

Wei, G., Zhou, Z., Guo, Y., Dong, Y., Dang, H., Wang, Y. and Ma, J. 2014. Long-term effects of tillage on soil aggregates and the distribution of soil organic carbon, total nitrogen, and other nutrients in aggregates on the semi-arid loess plateau, China. *Arid Land Research and Management*, *28*(3), pp.291-310.

Weltz, M.A., Kidwell, M.R. and Fox, H.D. 1998. Influence of abiotic and biotic factors in measuring and modeling soil erosion on rangelands: state of knowledge. *Range land Ecology and Management/Journal of Range Management Archives*, *51*(5), pp.482-495.

Wheater, H.S., Tuck, S., Ferrier, R.C., Jenkins, A., Kleissen, F.M., Walker, T.A.B. and Beck, M.B. 1993. Hydrological flow paths at the Allt a'Mharcaidh catchment: an analysis of plot and catchment scale observations. *Hydrological Processes*, *7*(4), pp.359-371.

Whitlow, R., 1986. Mapping erosion in Zimbabwe: a methodology for rapid survey using aerial photographs. *Applied Geography*, *6*(2), pp.149-162.

Williams, F. M. 2016. Understanding Ethiopia Geology and Scenery. *Springer International Publishing, Switzerland.*

Wilson, G.V., Cullum, R.F. and Römkens, M.J.M. 2008. Ephemeral gully erosion by preferential flow through a discontinuous soil-pipe. *Catena*, *73*(1), pp.98-106.

Wischmeier, W.H. and Smith, D.D. 1978. *Predicting rainfall erosion losses: a guide to conservation planning* (No. 537). Department of Agriculture, Science and Education Administration.

Woldemariam, G. and Harka, A. 2020. Effect of land use and land cover change on soil erosion in Erer Sub-Basin, Wabi Shebelle Basin, North east Ethiopia. *Land*, 9(4), p.111.

Wolka, K., Tadesse, H., Garedew, E. and Yimer, F., 2015. Soil erosion risk assessment in the Chaleleka wetland watershed, Central Rift Valley of Ethiopia. *Environmental Systems Research*, *4*(1), pp.1-12.

Wu, Q.S., Srivastava, A.K., Cao, M.Q. and Wang, J. 2015. Mycorrhizal function on soil aggregate stability in root zone and root-free hyphae zone of trifoliate orange. *Archives of Agronomy and Soil Science*, *61*(6), pp.813-825.

Yalew, S.G., Mul, M.L., Van Griensven, A., Teferi, E., Priess, J., Schweitzer, C. and van Der Zaag, P. 2016. Land-use change modelling in the Upper Blue Nile Basin. *Environments*, *3*(3), p.21.

Yan, X., Li, J., Shao, Y., Hu, Z., Yang, Z., Yin, S. and Cui, L. 2020. Driving forces of grassland vegetation changes in Chen Barag Banner, Inner Mongolia. *GIScience & Remote Sensing*, *57*(6), pp.753-769.

Yesuph, A.Y. and Dagnew, A.B. 2019. Soil erosion mapping and severity analysis based on RUSLE model and local perception in the Beshillo Catchment of the Blue Nile Basin, Ethiopia. *Environmental Systems Research*, *8*(1), pp.1-21.

Yimer, F., Ledin, S. and Abdelkadir, A. 2006. Soil organic carbon and total nitrogen stocks as affected by topographic aspect and vegetation in the Bale Mountains, Ethiopia. *Geoderma*, *135*, pp.335-344.

Yitaferu, B. 2007. Land degradation and options for sustainable land management in the Lake Tana Basin (LTB), Amhara Region, Ethiopia (Doctoral dissertation, University of Bern).

Yoseph, I., Delelegn, T., Witoon, P., Amila, B., Birru,Y., Tesfaye, W., Hans, G. and Douglas, L. 2017. Changes in land use alter soil quality and aggregate stability in the highlands of northern Ethiopia. Scientific Reports.7: 13602.

Young, N.E., Anderson, R.S., Chignell, S.M., Vorster, A.G., Lawrence, R. and Evangelista, P.H. 2017. A survival guide to Landsat preprocessing. *Ecology*, *98*(4), pp.920-932.

Yu, Q., Hu, X., Ma, J., Ye, J., Sun, W., Wang, Q. and Lin, H. 2020. Effects of long-term organic material applications on soil carbon and nitrogen fractions in paddy fields. *Soil and Tillage Research*, *196*, p.104483.

Zeleke, G. and Hurni, H. 2001. Implications of Land Use and Land Cover Dynamics for Mountain Resource Degradation in the Northwestern Ethiopian Highlands. *Mountain Research and Development,* 21(2): 184–191.

Zemadim, B., McCartney, M., Langan, S. and Sharma, B., 2014. *A participatory approach for hydrometeorological monitoring in the Blue Nile River Basin of Ethiopia* (Vol. 155). IWMI.

Zerihun, M., Mohammedyasin, M.S., Sewnet, D., Adem, A.A. and Lakew, M. 2018. Assessment of soil erosion using RUSLE, GIS and remote sensing in NW Ethiopia. *Geoderma regional*, *12*, pp.83-90.

Zhang, X., Davidson, E.A., Zou, T., Lassaletta, L., Quan, Z., Li, T. and Zhang, W., 2020. Quantifying nutrient budgets for sustainable nutrient management. *Global Biogeochemical Cycles*, *34*(3), p.e2018GB006060.

Zhang, Z. and He, G., 2013. Generation of Landsat surface temperature product for China, 2000–2010. *International journal of remote sensing*, *34*(20), pp.7369-7375.

Zhang, Y., Odeh, I.O. and Han, C., 2009. Bi-temporal characterization of land surface temperature in relation to impervious surface area, NDVI and NDBI, using a sub-pixel image analysis. *International Journal of Applied Earth Observation and Geoinformation*, *11*(4), pp.256-264.

Zhao, D., Arshad, M., Li, N. and Triantafilis, J. 2021. Predicting soil physical and chemical properties using vis-NIR in Australian cotton areas. *Catena*, *196*, p.104938.

Zingore, S., Mutegi, J., Agesa, B., Tamene, L. and Kihara, J. 2015. Soil degradation in sub-Saharan Africa and crop production options for soil rehabilitation. *Better Crops*, *99*(1), pp.24-26.

Zornoza, R., Acosta, J.A., Bastida, F., Domínguez, S.G., Toledo, D.M. and Faz, A. 2015. Identification of sensitive indicators to assess the interrelationship between soil quality, management practices and human health. *Soil*, *1*(1), pp.173-185.

List of published articles in reputable journals

The following articles wren published in reputable journals

Environmental Challenges 7 (2022) 100523

Contents lists available at ScienceDirect

Environmental Challenges

journal homepage: www.elsevier.com/locate/envc

Detection of land use/land cover and land surface temperature change in the Suha Watershed, North-Western highlands of Ethiopia

Nigussie Yeneneh[a,b,*], Eyasu Elias[a], Gudina Legese Feyisa[a]

[a] College of Natural and Computational Science, Addis Ababa University, Addis Ababa, Ethiopia
[b] College of Agriculture and Natural Resource Management, Debre Markos University, Debre Markos, Ethiopia

ARTICLE INFO

Keywords:
LU/LC change
LST, GIS & remote sensing
Landsat image
Suha watershed

ABSTRACT

Human-induced land use land cover changes resulted in adverse impacts on the environment at various spatial and temporal scales. The Highland regions of Ethiopia are typical examples of these phenomena. The objective of this study was to analyze spatiotemporal changes in land use/ land cover and their impacts on land surface temperature in the suha watershed, northwestern highlands of Ethiopia. Multi-temporal Landsat images (1985–2019) were used to analyze LU/LC changes and LST using GIS and remote sensing techniques. Image preprocessing, supervised classification, accuracy assessment, and change detection were conducted to identify LU/LC classes, area coverage, and their transitions. Thermal bands of these satellite images were also used to extract LST.

Significant changes in land use/land cover (spatial and temporal) were observed in the suha watershed during the study periods. Agricultural land has got the largest proportion in all study periods. The barren land and built area also expanded greatly in 35 years period. Agricultural land has increased by 15417.6 ha (34.8%) and bare land has expanded by 5297.2 ha (373.6%). However, grazing and shrub lands were reduced by18568.4 ha (72.1%) and 3544.2 ha (47.6%), respectively. Spatial and temporal variation of LST was also observed in the same period. The highest mean values were found on impervious surfaces (built-up areas and bare land), and the lowest values were recorded on forest land. A negative correlation was found between LST and NDVI. Undesirable land use and land cover changes put greater pressure on environmental resources, resulting in an adverse effect on them. Therefore, to reverse this situation and create a balanced ecosystem, management strategies should be applied that mainly focus on soil conservation technologies in steep slope areas, improve afforestation and apply proper land-use policies. The outcomes of this research are useful in designing and implementing appropriate strategies that can address critical social and environmental problems. It also provides new knowledge that helps us better understand the spatiotemporal land use land cover changes and their impacts on LST.

Yeneneh et al.
Environmental Systems Research (2022) 11:20
https://doi.org/10.1186/s40068-022-00265-5

Environmental Systems Research

RESEARCH **Open Access**

Quantify soil erosion and sediment export in response to land use/cover change in the Suha watershed, northwestern highlands of Ethiopia: implications for watershed management

Nigussie Yeneneh[1,2*], Eyasu Elias[1] and Gudina Legese Feyisa[1]

Abstract

Soil erosion accelerated by human activities is a critical challenge affecting soil health, agricultural productivity, food security and environmental sustainability in the highlands of Ethiopia. The aim of this study was to examine the dynamics of soil loss and sediment yield potential, and identify soil erosion hotspots using RUSLE with GIS in the Suha watershed, north western highlands of Ethiopia. Digital Elevation Model, LU/LC, rainfall, soil, and conservation practice were used as input data for RUSLE model. The estimated total annual soil loss for the entire watershed increased from 1.22 million tons in 1985 to 2.43 million tons in 2019, with average annual soil loss rates of 15.2 t ha^{-1} yr^{-1} and 31.4 t ha−1 yr^{-1} respectively. Total sediment yield also increased from 317.52 to 630.85 thousand tons over the past 35 years. In addition, the area of soil erosion hotspots changed from 15.2% (12,708 ha) to 32% (25,660 ha) during the same periods. Sub watershed 1, 2, 15, 17, 18, and 23 are severely degraded parts of the watershed. Expansion of agriculture and bare land as the expenses of other land use types over the past 35 years could be the major causes of extensive soil erosion risk in the watershed. Besides its temporal variability, soil loss and sediment export also showed variation between land use/cover classes. The estimated results of soil loss and sediment yield as well as soil erosion hotspots revealed that the soil erosion risk is progressively increasing during the study periods. Unless action is taken and the current condition is reversed, it will critically threat the livelihoods of the community in the watershed. Generally, the results underscore urgent demand for integrated and effective watershed management strategies.

Keywords: LULC change, GIS, RUSLE, Soil erosion, SY, Suha watershed

ACTA AGRICULTURAE SCANDINAVICA, SECTION B — SOIL & PLANT SCIENCE
2022, VOL. 72, NO. 1, 1009–1019
https://doi.org/10.1080/09064710.2022.2142658

Taylor & Francis
Taylor & Francis Group

OPEN ACCESS Check for updates

Assessment of the spatial variability of selected soil chemical properties using geostatistical analysis in the north-western highlands of Ethiopia

Nigussie Yeneneh[a,b], Eyasu Elias[a] and Gudina Legese Feyisa[a]

[a]College of Natural and Computational Science, Addis Ababa University, Addis Ababa, Ethiopia; [b]College of Agriculture and Natural Resource Management, Debre Markos University, DebreMarkos, Ethiopia

ABSTRACT

The aim of this research was to evaluate the status of selected soil chemical properties and to explore their spatial variability in various agroecosystems (AEs) of Choke Mountain, northwestern highlands of Ethiopia. A total of 74 geo-referenced composite surface (0-30 cm) soil samples were collected from representative farm lands. Soil chemical properties were analyzed following standard procedures. In addition, semi-variogram model was applied to estimate the spatial variability of soil properties. The results showed that soil chemical properties are below their threshold values in all agroecosystems and soil groups. The value of soil pH ranged from 4.3 to 6.8; Exchangeable acidity ranged from 0.124 to 0.180 cmol (+) kg^{-1}; EC ranged from 0.02 to 0.19 dS m^{-1}; OC ranged from 0.7 to 3.8%; TN ranged from 0.11 to 0.27%; Exchangeable potassium from 0.29 to 2.0 cmol (+) $kg^{-1}$and CEC from 22 to 56 cmol (+)kg^{-1}. The results also revealed that soil pH, exchangeable acidity, electrical conductivity, and exchangeable potassium showed strong spatial dependence whereas SOC and TN had moderate spatial dependence. The results indicated that soil quality is deteriorating. Hence, crop, soil and site-specific soil management strategies, primarily focusing on improving the content of SOM and addressing the problems of soil acidity are suggested to improve the soil conditions.

Abbreviations: AES, agroecosystem; CEC, cation exchange capacity; EC, electrical conductivity; Ex.K, exchangeable potassium; OK, ordinary Kriging; OM, organic matter; TN, total nitrogen.

ARTICLE HISTORY
Received 6 June 2022
Accepted 28 October 2022

KEYWORDS
Agroecosystem; spatial variability; geo statistics; soil chemical properties; soil groups

Yeneneh et al.
Environmental Systems Research (2024) 13:7
https://doi.org/10.1186/s40068-024-00336-9

Environmental Systems Research

RESEARCH **Open Access**

Monitoring soil quality of different land use systems: a case study in Suha watershed, northwestern highlands of Ethiopia

Nigussie Yeneneh[1,2*], Eyasu Elias[1] and Gudina Legese Feyisa[1]

Abstract

The problem of soil quality degradation has been becoming more severe in the highlands of Ethiopia due to soil erosion; land use and land cover change, and poor land management. The level of soil quality degradation was not well known and documented in the study area and the results of this study could provide new information to improve soil conditions. The present study was conducted to evaluate soil quality in terms of its physical and chemical fertility under different land use types in the Suha watershed, northwestern highlands of Ethiopia. A total of 27 composite surface soil samples (0–30 cm) were collected from adjacently located land-uses in three replications from two elevation gradients. Standard procedures were followed to analyze selected soil physical and chemical quality indicators. The differences in the mean values of the parameters were tested using a two-way analysis of variance. In addition, Soil Quality Degradation Index was evaluated to see the direction and magnitude of change in soil quality indicators. The analysis of variance results revealed that soil quality indicators such as index of soil aggregate stability, organic carbon (OC), total nitrogen (TN), and C:N ratio were significantly decreased in the cultivated land use system compared to other land use systems. On the other hand, the content of available Phosphorus was significantly higher in the cultivated land. Soil quality deterioration index values were highly negative for SOC (−71.3%) and TN (−67.7%) in the cultivated land, followed by grazing land (SOM = −35.5% and TN = −27.7%). Aggregated Soil Quality Index values also indicated that the status of soil quality under cultivated fields is rated as low, grazing land as optimal, and forest land as high. Generally, results indicated that land use and cover changes had adverse effects on soil quality indicators. Hence, soil management strategies, mainly Integrated Soil Fertility Management which integrates soil and water conservation strategies, are required to alleviate the problem of soil quality deterioration and improve agricultural productivity.

Keywords Soil quality, Land use type, Soil Quality Index, Suha watershed

Reviewed manuscript waiting for final decision

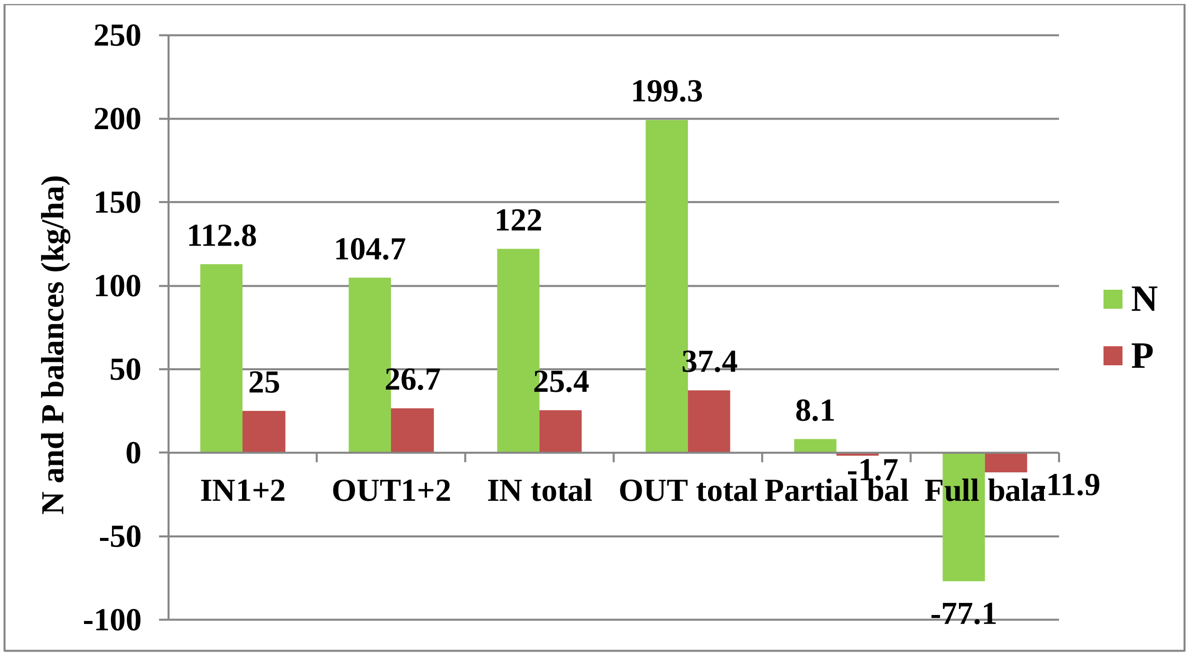
5. Nigussie Yeneneh, Eyasu Elias & Gudina Legese Feyisa. 2024. **Assessment of soil nutrient flows and balances in cereal-based agroecosystems of Suha watershed, Northwestern Highlands of Ethiopia: Implications for the sustainability of soil fertility management.** *Journal of Environment, Development and Sustainability.*

APPENDICES

Appendix 1. Mean values of soil physical and chemical properties under different LUTs along the toposequence of the watershed

Land use types	Soil properties																	
	Sand	Clay	Silt	Textural class	FC	PWP	WHC	PH-H2O	OC	TN	C:N	Ap	NA	K	CA	MG	CEC	PBS
		%			% (0.33 bar)	% (15 bar)	%		%	%		mg/kg			cmol(+)/kg			%
Upper part of the watershed																		
Cultivated land	2.6	72.7	24.8		49.0	33.6	15.4	5.8	1.0	0.1	11.2	45.0	1.3	0.8	33.3	13.3	64.0	75.0
Forest land	35.1	19.9	45.0		41.8	29.1	12.7	6.0	3.9	0.2	18.7	27.8	1.2	2.0	39.0	11.1	69.1	77.0
Grazing land	15.2	49.9	34.9		50.7	35.2	15.5	5.6	2.0	0.1	15.9	18.6	1.2	0.6	42.1	15.9	69.9	82.2
Middle part of the watershed																		
Cultivated land	14.8	54.2	31.0		41.2	28.3	12.9	5.6	1.1	0.1	14.4	27.8	1.5	0.8	28.8	10.4	59.6	70.3
Forest land	3.2	61.0	35.8		48.6	34.8	13.8	5.6	3.0	0.2	18.6	13.1	1.5	1.0	40.0	14.4	72.7	78.4
Grazing land	14.3	51.5	34.1		46.6	32.6	14.1	5.8	2.5	0.2	13.8	14.9	1.2	1.7	33.5	13.9	66.2	75.1
Lower part of the watershed																		
Cultivated land	15.5	56.5	28.0		60.1	42.2	17.9	6.1	1.1	0.1	13.4	54.2	0.9	0.9	52.0	21.4	74.0	103.1
Forest land	10.7	61.2	28.1		51.7	36.0	15.7	5.8	3.1	0.2	15.4	23.0	2.0	1.1	41.2	13.2	80.4	72.9
Grazing land	7.6	56.5	35.9		47.4	34.0	13.3	5.6	2.7	0.2	14.6	20.3	1.4	0.8	40.1	14.0	69.1	79.3

Appendix fig1. Nutrient flows, partial and full balances of N and P in the highland part of the watershed (wheat farming system)



Appendix 2. Continued. Nutrient flows, partial and full balances of N and P in the mid land part of the watershed (teff farming system).