UNIVERSITY OF ZAMBIA, IWRM CENTER
SCHOOL OF MINES

QUANTITATIVE ANALYSIS OF THE IMPACT OF DEFORESTATION ON THE ECOSYSTEM SERVICES IN KAMFINSA SUB-CATCHMENT OF KITWE

By
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A dissertation submitted to the University of Zambia in partial fulfilment of requirements for the award of the Master of Science by Research in Integrated Water Resource Management

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DECLARATION

I declare that the work I am submitting for assessment contains no section copied in whole or in part from any other source unless explicitly identified in quotation marks and with detailed, complete and accurate referencing

APPROVAL

This Dissertation of Deuteronomy Kasaro is approved as fulfilling the requirement of the Degree of Masters of Science by Research in Integrated Water Resources Management by the University of Zambia.

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LIST OF ABBREVIATIONS

AGB	Above Ground Biomass
ANPP	Annual Net Primary Productivity
CBD	Conversion of Biological Diversity
BGB	Below Ground Biomass
CO_2	Carbon dioxide
DRC	Democratic Republic of Congo
ET	Evapotranspiration
EVI	Enhanced Vegetation Index
FAO	Food and Agriculture Organization of the United Nations
DEM	Digital Elevation Model
GDP	Gross Domestic Product
GIS	Geographical Information System
GHG	Green House Gases
GRZ	Government of the Republic of Zambia
HWSD	Harmonised World Soil Database
IWRM	Integrated Water Resource Management
MAI	Mean Annual Increment
MEA	Millennium Ecosystem Assessment
NIR	Near Infrared
NDVI	Normalised Difference Vegetation Index
NKP	Nitrogen, Potassium, and Phosphorus (three essential plant nutrients).
OM	Organic Matter
pH	Potential of Hydrogen (a numeric scale used to specify the acidity or basicity of an aqueous solution)
REDD+	Reducing Emissions from Deforestation and forest Degradation and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries
RUSLE	Revised Universal Soil Erosion Equation
RS	Remote Sensing
SAGA	System for Automated Geoscientific Analyses (Software)
UNCCD	United Nations Conversion to Combat Desertification
USLE	Universal Soil Loss Equation
IIED	International Institute for Environment and Development
IUCN	International Union for Conservation of Nature and Natural Resources

ABSTRACT

The objective of the study was to provide a holistic evaluation of the impacts of deforestation on ecosystem services in Kamfinsa sub-catchment of Kitwe in Zambia. Loss of forest biomass, biodiversity, and soil erosion were key proxy variables for loss of ecosystem services. Geographical Information System (GIS) and Remote sensing techniques were used to assess the impacts on the ecosystem. The results showed that deforestation reached 576.3 hectares per year. The forest loss corresponds to an emission of 43.73 ton of carbon per hectare from above and below ground biomass valued at US$243.60 per hectare. According to a soil erosion risk assessment, 1.59 ton of soil was lost per hectare per year, equivalent to US$ 57.20 loss per hectare. The indigenous forest cover reduced from 13,430.5 ha (1990) to 2,904.7 ha (2010), with a corresponding change in NDVI index for loss of forest vigor and biodiversity of 0.56 and 0.32, respectively. The major forest loss occurred from indigenous forests. The study has shown that deforestation in Kamfinsa sub-catchment area calls for the urgent promotion of an integrated and comprehensive approach to addressing the drivers of deforestation to ensure continued supply of ecosystem services.

1 INTRODUCTION

Ecosystems depend on fundamental environmental cycles such as the continuous circulation of water, carbon, and other nutrients. The water, food and raw materials needed for human livelihood security originate from the ecosystems surrounding human settlements (Falkenmark, 2003). According to Evans, (2005), human activities have modified these cycles, especially during the last 50 years, through increases in freshwater use, carbon dioxide emissions, and fertilizer use, which has affected ecosystem's ability to provide benefits to humans. Historically, the nature and value of Earth's life support systems have largely been ignored until their disruption or loss highlighted their importance (Evans, 2005). The removal of all forests, for example, would involve the loss of a major life support system (CBD, 2001).

While there has been much interest in valuing the totality of ecosystem services (Constanza, et al., 1997), such exercises have no economic meaning in this context because the question as to what is the 'value of everything' has no meaning (Pearce & Pearce, 2001). The appropriate context for economic valuation is, therefore, the value of a small or a discrete change in the provision of goods and services through, say, the loss or gain of a given increment or decrement in forest cover (CBD, 2001). The values of forests, therefore, embody the values of the biological diversity they contain since it seems unlikely that the vast majority of the biological resources in question could occupy non-forest habitats (CBD, 2001)

Deforestation has recently revealed the critical role forests serve in regulating the hydrological cycle, in particular, in mitigating floods, droughts, the erosive forces of wind and rain, and silting of dams and irrigation canals. The primary threats are land use changes that cause losses in biodiversity as well as disruption of carbon, nitrogen, and other biogeochemical cycles; human-caused invasions of exotic species; releases of toxic substances; possible rapid climate change; and depletion of stratospheric ozone (Daily, 2000).

Since ecosystems are genuinely water-dependent, it is becoming essential that the linkages between water and ecosystems be adequately clarified. However guidance from the literature on the linkages between hydrology and ecology is, unfortunately, limited (Falkenmark, 2003).

In recognizing that Agriculturists or farmers are key managers of most productive lands on earth, (Dale, 2007) this study quantified the impacts of human activity on ecosystems services by focusing on agricultural practices and encroachment. The approach aimed at enhancing the understanding of the linkages of environment, economic and social dimensions. Therefore, since both humans and ecosystems are genuinely water dependent, Integrated Water Resource Management (IWRM) offers an opportunity to take an integrated approach to human livelihood security and protection of vital ecosystems.

According to the (UNCCD, 2012), land degradation is a reduction or loss, in arid, semi-arid and dry sub humid areas of biological or economic productivity or complexity of rain fed cropland, irrigated cropland, or range, pasture, forest and woodlands resulting from land uses or from a process or combination of processes, including process arising from human activities and habitation patterns, such as soil erosion caused by wind and/or water; deterioration of the physical, chemical and biological or economic properties of; and long term loss of natural vegetation.

Degradation of natural resources, particularly land and forests, has become a serious concern in developing countries where most rural people depend on these resources for sustenance (FAO, 2011). Deforestation and inappropriate agricultural practices have undermined the productivity capacity of approximately two billion hectares (ha) of the world's agricultural land (Rasul, 2009). Land use change, including conversion of forest land into agricultural land, not only accelerate land degradation but also intensifies carbon-dioxide (CO_2) emission and loss of biological resources (Jackson, et al., 2008).

One of the primary causes of global environmental change is tropical deforestation, but the question of what factors drive deforestation remains largely unanswered (Geist & Lambin, 2001). Agricultural expansion is by far, the leading land use change. There are proximate and underlying drivers of deforestation, which need to be addressed (Figure 1).

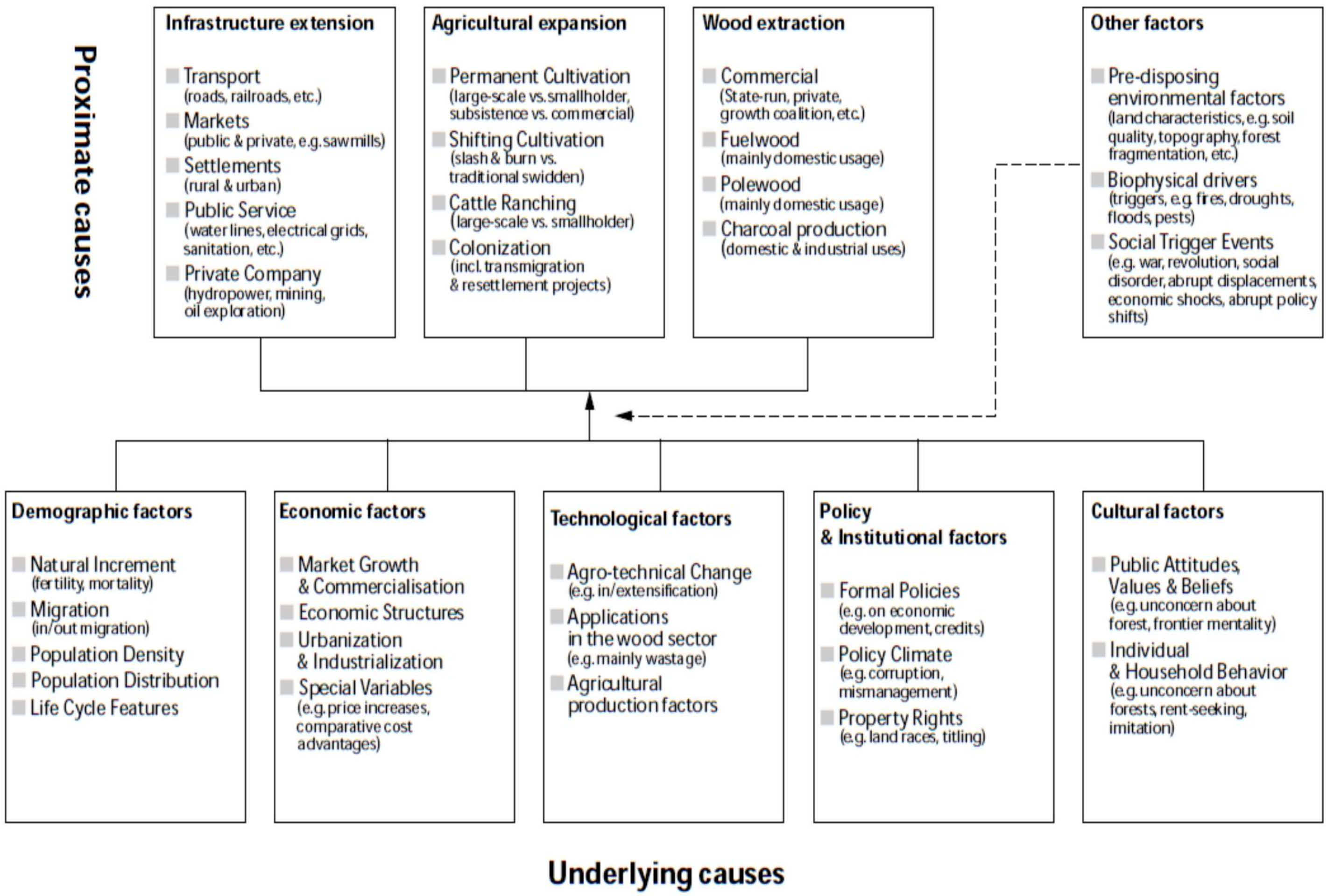

Figure 1: Drivers of Deforestation (Ecosystem Degradation). Geist and Lambin, (2002)

1.1.1 Biodiversity and Ecosystems

Biodiversity (the number, abundance, composition, spatial distribution and interactions of genotypes, populations, species, functional types and traits, and landscape units) in a given system contributes to human well-being through its effects on the ecosystem processes that lie at the core of the Earth's vital life support systems (Falkenmark, 2003). Different components of biodiversity (species richness, genetic richness, kind, abundance and range of functional traits) affect different ecosystem processes and services to different degrees (Díaz, et al., 2006). Most of the information about positive effects of biodiversity on ecosystem processes is at the level of species richness (MEA, 2005). Ecosystem services are the benefits people obtain from the ecosystem.

1.1.2 Forest and Water Relations

According to (Falkenmark, 2003), terminology of "green" and "blue water", green water is the return of water to the atmosphere as evapotranspiration (ET), including transpiration by vegetation, evaporation from soil, lakes and water intercepted and evaporated from (mainly tree canopy surfaces) i.e to a large part of water that is used to produce food and environmental services by forests and agricultural crops. Blue water is, on the other hand, what is left contributes to ground water and stream runoff, i.e., water available for animal and human consumption for example in downstream areas. Critical processes are the partitioning of rainfall between green and blue water which is:

a. Infiltration of water into soil or surface runoff and
b. Uptake of soil water by plants or recharge of ground water.

It is an empirically and theoretically well-established scientific paradigm that forests use more water than lower vegetation and rain-fed agriculture. This could also be expressed as a positive relationship between biomass production and water use (Rockstrom, et al., 2010). Consequently, the empiric evidence is strong that cutting forests results in increased stream flow (Bosch & Hewlett, 1982). Typically, also regenerating forests and afforestation is shown to partition more of the rainfall to green water reducing availability of blue water (Bremer & Farley, 2010).

Forests have been shown to maintain high infiltrability by superior litter fall, more large pores, root channels and soil protection against surface runoff. So in these aspects we have some evidence than "sponge effect" can be lost by deforestation and subsequent soil degradation, but the conclusion can hardly be made general for all semi-arid forest ecosystems (Malmer & Nyberg, 2008).

Increasing surface runoff after deforestation and possible soil deterioration leads to more "blue water" in streams momentarily (Falkenmark, 2003). Land cover and use directly impact the amount of evaporation, groundwater recharge and overland runoff that occurs during and after precipitation events. These factors control the water flow yields of surface streams and groundwater aquifers and thus the amount of water available for both ecosystem function and human use (Mustard & Fisher, 2004). Changes in land cover and use alter both runoff behavior and the balance that exists between evaporation, groundwater recharge and stream discharge in specific areas and entire watersheds, with considerable consequence for all water users (Sahin & Hall, 1996)

Soils under Miombo woodlands are primarily associated with deep, colluvial, stoneless sandy loams or sandy clay occurring on geologically old and acidic (pH 4-6) soils with inherently low fertility. These soils often contain laterite nodules and mica aggregates in the B horizon. Organic material in these soils is a very important component of affecting soil quality and fertility. Surface soil under Miombo woodland tend to have high organic matter however upon exposure to cultivation they are reduced up to 50% (Malmer & Nyberg, 2008). Soil organic matter strongly affects soil physical properties. The soil structure (soil aggregates increase amount of large pores) determines to a large extent the partitioning of rainfall into surface runoff, erosion and soil infiltrability (Bruijnzeel, 1990). In various land uses in Zambia, structure stability of the soil show direct and positive correlation to soil organic carbon (King & Campbell, 1993). According to Hough, (1986) many problems identified under semi-arid areas landscapes are;

a. Increasing day season flow;
b. Tree preserve infiltrability and reduction surface runoff and erosion but;
c. Trees forest use more water than other vegetation.

According to King and Campbell (1993), grasses may have considerable biomass production potential, but they have less deep roots than trees. Miombo Conversion to grass consumes less water and increase stream flow. In contrast, grasses after deforestation often show lower infiltrability and thereby increase surface runoff, erosion and storm flow (Falkenmark and Rockstorm, 2008), stress the importance of increasing efficiency in cultivated systems to shift losses in evaporation to productive transpiration.

The Kamfinsa sub-catchment areas are covered by Acrisols, (Figure 2). These soils are under conditions of high leaching intensity. The soils are characterized by soil acidity, low bases retention capacity, low soil organic matter, low soil fertility and soil degradation. They are characterized by the accumulation of low activity clays in the subsurface horizon and by a low base saturation level. These soils have a subsurface horizon with distinct higher clay content than the overlying horizon, which has a cation exchange capacity of less than 24 cmolc per kg in some part, either starting within 100 cm from the soil surface. They have a base saturation (total amount of Ca, Mg, K and Na with respect to the cation exchange capacity) of less than 50 percent in the major part between 25 and 100 cm from the soil surface.

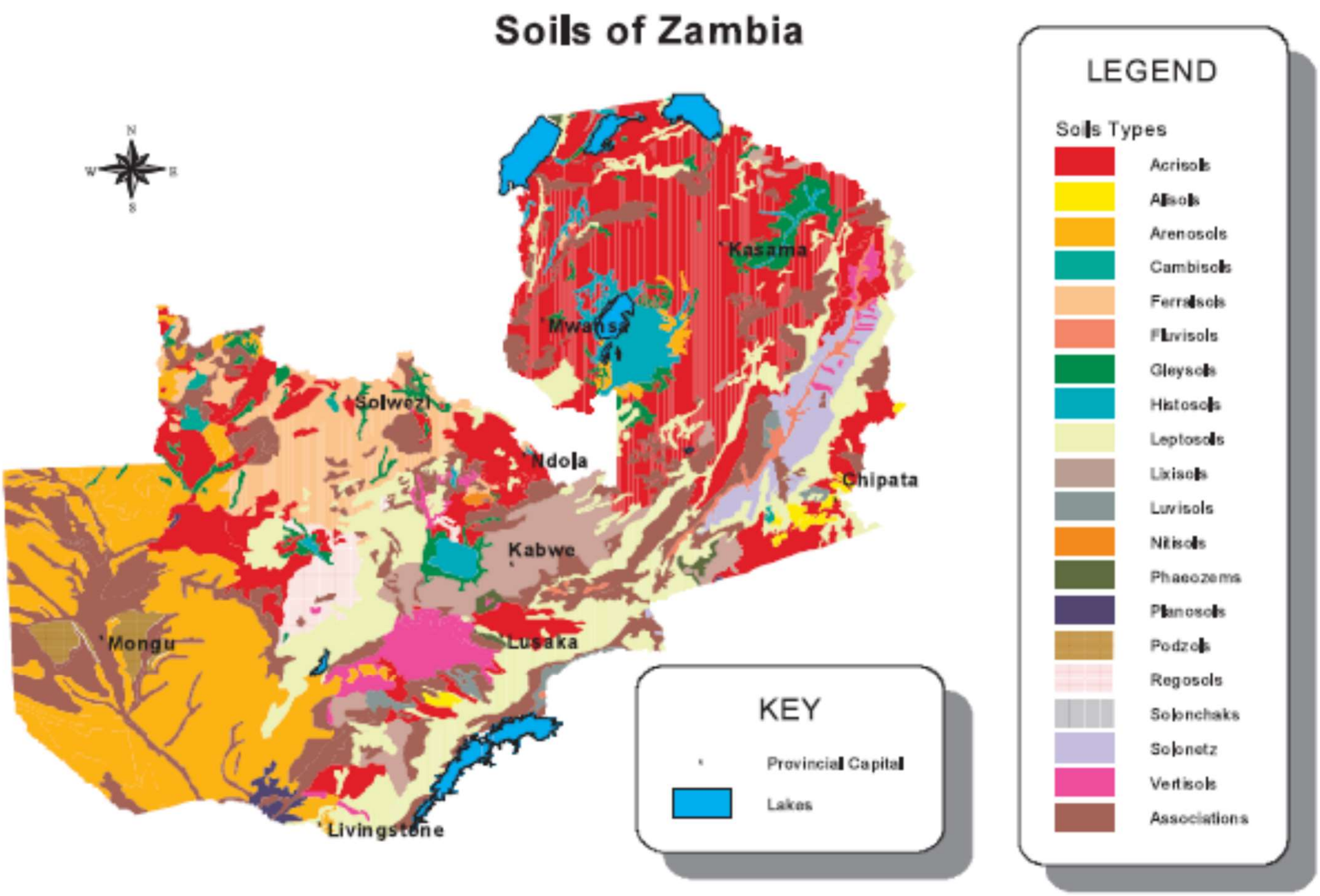

Figure 2: Soil Map of Zambia

1.1.3 Miombo Woodlands

The Miombo Ecoregion covers over 3.8 million km2 in central and southern Africa, extending from the west coast in Angola to the east coast in Mozambique and Tanzania (Timberlake & Chidumayo, 2001). It includes all or part of 11 countries – Angola, Namibia, Botswana, South Africa, Zimbabwe, Zambia, Democratic Republic of Congo (DRC), Mozambique, Malawi, Tanzania and Burundi, (Figure 3). According to Timberlake and Chidumayo, 2011, much of the ecoregion is on the ancient African plateau with an altitude of 800 to 1250m above sea level, but in the east the ecoregion transcends the escarpment and elements of the ecoregion can be found in the east African coastal zone, at 200 to 300 m altitude.

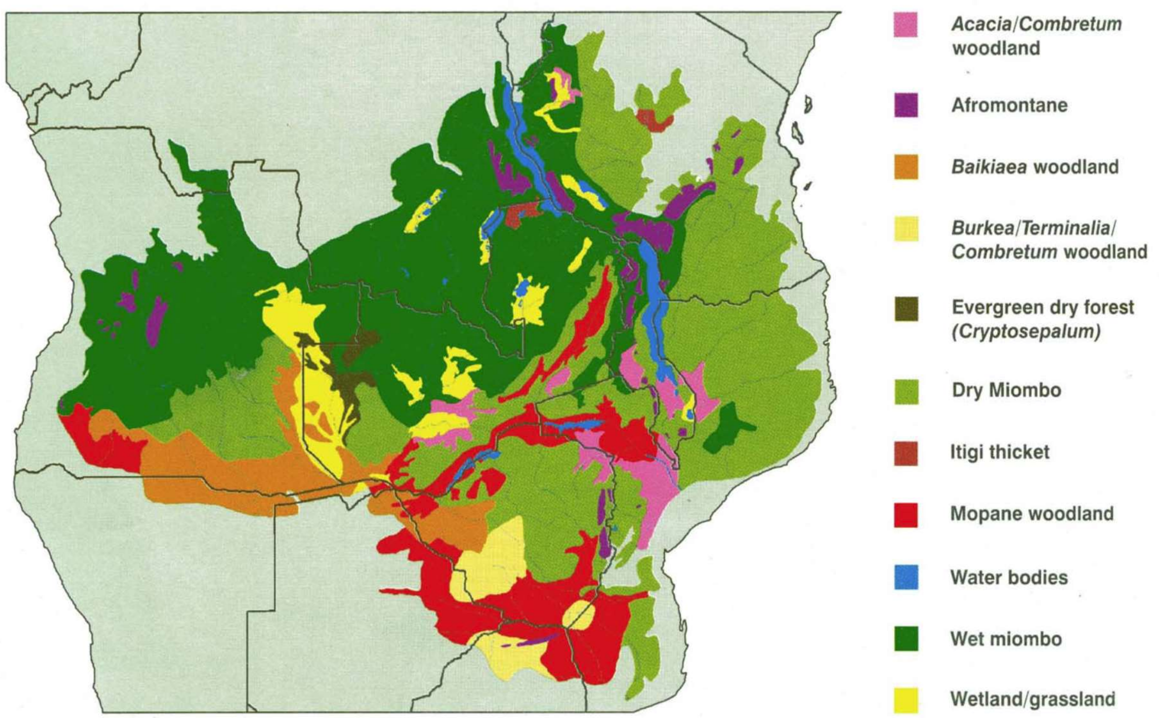

Figure 3 Miombo Ecoregion - Vegetation (from WWF SAPRO 2003)

Despite the favourable elevation, the biological elements of the ecoregion give way to other biomes in the northeast, south, and southwest. Elevation, therefore, does not fully determine the Miombo Ecoregion boundary (Timberlake & Chidumayo, 2001)

Mature, relatively undisturbed stands typically comprise a 10-20 m high, single storey, partly closed canopy of mostly pinnate-leafed trees; a discontinuous understorey of broad-leafed shrubs; and an often sparse but continuous herbaceous layer of forbs, small sedges, and caespitose, heliophytic C4 grasses (Campbel, et al., 1996). The ecoregion is incised by the large river valleys of the Zambezi, Luangwa and Limpopo, and by the Rift Valley lakes of Tanganyika and Malawi. A number of major drainage basins such as the Zambezi, Limpopo, Save, Cuando, Kavango, Rufiji, Rovuma and Luapula (part of the Upper Congo) are incorporated (Timberlake & Chidumayo, 2001).

1.2 Research Problem

The problem of continued fragmented approach to environment and natural resources management poses challenges in sustainable ecosystem management leading to deforestation and land degradation. There is need to understand the inter-linkages and the cost of land degradation in order to enhance planning for ecosystem services management.

1.3 Justification

One of the important causes for widespread, continuing land degradation is that the various economic benefits that are provided by multifunctional agricultural landscapes and natural ecosystems tend to be underestimated in decision making (Hein, 2009). Knowledge of the nature of land use and land cover change and their configuration across spatial and temporal scales is consequently indispensable for sustainable environmental management and development (Turner, et al., 1995).

It is important therefore, to assess the impact of deforestation ecosystem services as it will provide information to enhance the planning process. A whole range of social, institutional, and economic factors play a role with regards to the lack of sustainability in the management of natural resources (Lambin, et al., 2001). The effect of structural adjustment programmes initiated in 1986 which were followed by a quick and massive closing and privatization of state-controlled industries in Zambia resulted in massive retrenchments and large numbers of urban unemployed people returned to the rural areas resulting in unplanned settlement and over-exploitation of forest resources in order to earn a living. (Jumbe, et al., 2009).

Hydrology has been the domain of engineers with a focus on river flow phenomena of societal relevance while ecology has been the domain of biologists with focus on climate/ecosystem linkages (Falkenmark, 2003). The inadequate governance associated with water resource development, particularly a single-minded, engineering – economic approach to the ecosystems services that inland water system provide, has led to significant social and environmental impacts; impacts that have disproportionately affected the rural poor that rely on the natural junction of water ecosystem (IIED, 2007)

Agriculture is a key driver of the global economy. However, the main challenge is to simultaneously secure enough high-quality agricultural production, conserve biodiversity and manage natural resources and improve human health and well-being, especially for the rural poor in developing countries (IUCN, 2008). The socio-economic and physical factors which drive soil erosion, therefore, need to be addressed in tandem (Boardman, et al., 2003).

Over the past two decades, tropical land-use change, especially deforestation and forest degradation, has accounted for 12–20% of global anthropogenic greenhouse gas (GHG) emissions (Chave, et al., 2014) Any national Reducing Emissions from Deforestation and forest Degradation (REDD+) scheme will need to have a reliable and credible system of measuring, reporting and verifying changes in forest carbon emissions reductions and removals (Atela, et al., 2016)

This study therefore aims at investigating the quantitative impacts of deforestation on ecosystem services in order to enhance the understanding of the linkages of social, economic and environmental issues that can be addressed through an appropriate ecosystem management approach which takes into account integrated water resource management. Therefore, can quantitative analysis of deforestation enhance ecosystem management planning?

1.4 Research Objectives

Overall objective

To quantify the impact of deforestation on ecosystem services to enhance planning for ecosystem services management.

Specific Objectives

i. To determine catchment deforestation (land cover change) over the past 20 years;

ii. To assess the loss of the proxy biophysical variables of ecosystem services (biomass, carbon, soil erosion and biodiversity); and

iii. To assess the cost of loss due to deforestation in the study area.

1.5 Research Question

Objective 1

i. How much land was converted from forest to another land use?
ii. What have been the main drivers of land use change in the study area?

Objective 2

i. How much soil erosion has occurred in the study area due to deforestation?
ii. How much carbon is lost due to deforestation in the study area?
iii. What has been an impact on biodiversity, using a proxy variable of Normalized Difference Vegetation Index (NDVI).

Objective 3

i. What are the costs of the loss of forests and soil erosion in the study area?

1.6 Research Hypothesis

Deforestation has a negative impact on ecosystem services.

1.7 Research Approach

The study focused on two broad aspects. Firstly, an assessment of land cover change from 1990 to 2010 mainly focusing on land-cover change, biomass loss and soil erosion. The ecological services of forests are many, and the values of forests, therefore, embody the values of the biological diversity they contain since it seems unlikely that the vast majority of the biological resources in question could occupy non-forest habitats (CBD, 2001). Secondly, using proxy variables, forest, biomass and soil erosion the study quantified the cost of deforestation. The quantitative assessment of the proxy variables provided an overview of the impacts of deforestation on ecosystem services (Fig. 4).

This provided a basis for justifying whether promoting integrated water resources management through ecosystem approach would enhance water resource management.

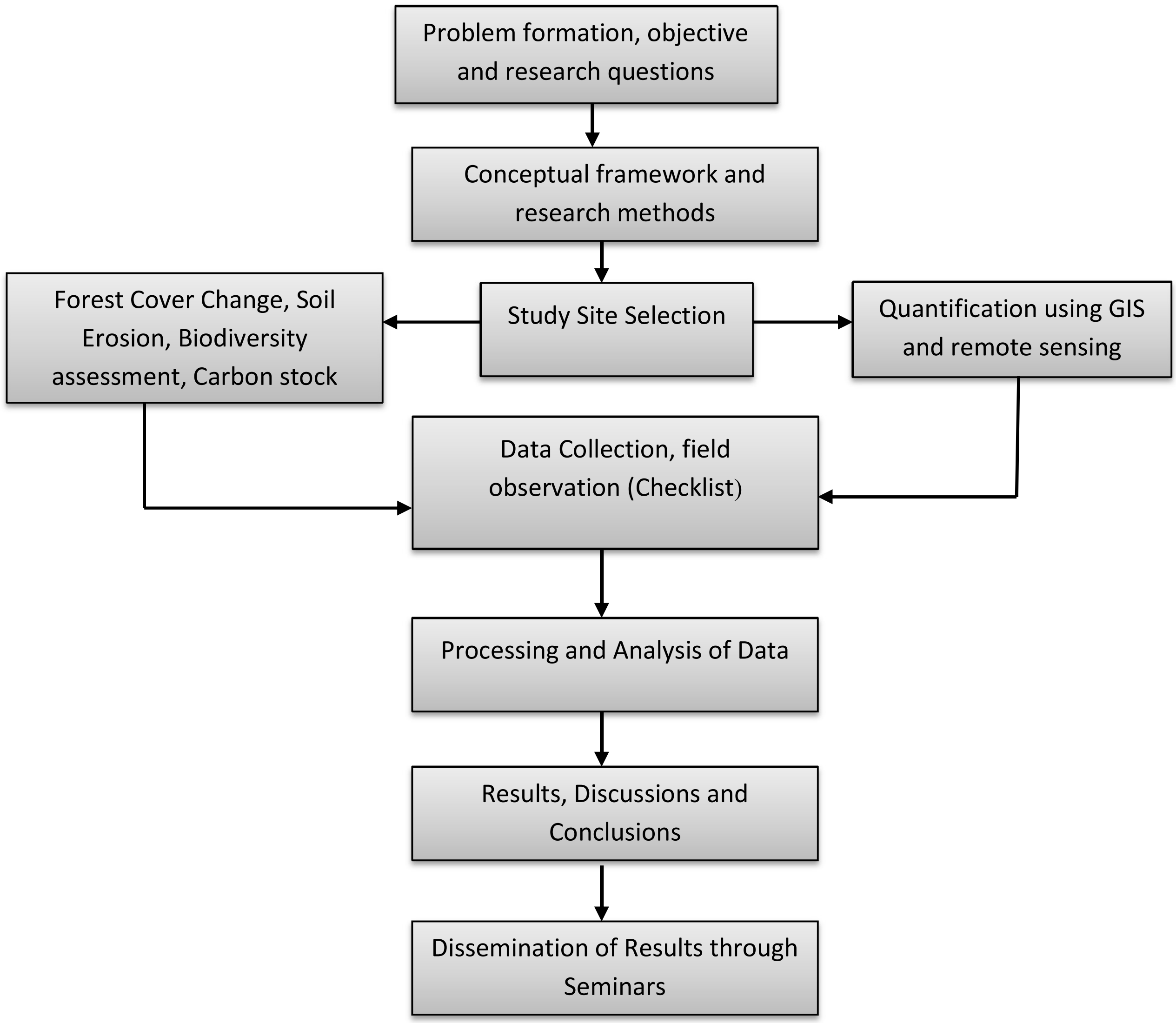

Figure 4: Research Conceptual Framework

1.8 The organisation of the Thesis

The Thesis has five Chapters.

- **Chapter 1,** gives a general introduction and addresses the main problem the research was meant to investigate.
- **Chapter 2,** reviews the literature on the importance of ecosystems and its degradation as well as the ecosystem services in the context of Zambia and the study area providing a wide view on which the research was built.
- **Chapter 3,** describes the research methods used to accomplish the study and describes the study area.
- **Chapter 4,** presents the results, analysis and discussions of the research study
- **Chapter 5,** concludes with the findings by reviewing the hypothesis and the findings.

2 LITERATURE REVIEW

Literature on ecosystem services provides a wide view of the various concepts and approaches to impacts of deforestation on ecosystem services. This chapter dwells on the importance of ecosystems and the type of ecosystem services available in the study area. Such arguments provide the foundation upon which the research was built. The presentation outlines the inter relationships of various factors in the ecosystem including loss of forest and soil.

Understanding the dynamics of land-use and land-cover has increasingly been recognized as one of the key research imperatives in global environmental change research (Geist & Lambin, 2001) and further says that since the 1980s, numerous attempts have been made to explain the causative pattern of tropical deforestation. Deforestation is complete removal of forest cover for another land use (FAO, 2005).

2.1 Ecosystem Services

According to the Millennium Ecosystem Assessment, 2005, an ecosystem is a dynamic complex of plant, animal, and microorganism communities and the nonliving environment interacting as a functional unit. In addition, the report further state that ecosystem services are the benefits people obtain from ecosystems, which include *provisioning services* such as food, water, timber, and fiber; *regulating services* that affect climate, floods, disease, wastes, and water quality; *cultural services* that provide recreational, aesthetic, and spiritual benefits; and *supporting services* such as soil formation, photosynthesis, and nutrient cycling (Table 1).

Changes in these services, affect human well-being through impacts on security, the necessary material for a good life, health, and social and cultural relations (Figure 2). These constituents of well-being are in turn influenced by and influence the freedoms and choices available to people (MEA, 2005)

While there is no single, agreed method of categorizing all ecosystem services, the Millenium Ecosystem Assessment framework is widely accepted and is seen as a useful starting point. (Defra, 2007).

Table 1: Millennium Assessment Categories of Ecosystem Services and Examples

ECOSYSTEM SERVICES	EXAMPLE OF ECOSYSTEM SERVICE
Provisioning services, i.e. products obtained from ecosystems	• Food, e.g. crops, fruit, fish • Fibre and fuel, e.g. timber, wool • Biochemicals, natural medicines, and pharmaceuticals • Genetic resources: genes and genetic information used for animal/plant breeding and biotechnology • Ornamental resources, e.g. shells, flowers
Regulating services, i.e. benefits obtained from the regulation of ecosystem processes	• Air-quality maintenance: ecosystems contribute chemicals to and extract chemicals from the atmosphere • Climate regulation, e.g. land cover can affect local temperature and precipitation; globally ecosystems affect greenhouse gas sequestration and emissions • Water regulation: ecosystems affect, e.g. the timing and magnitude of runoff, flooding, etc. • Erosion control: vegetative cover plays an important role in soil retention/prevention of land/asset erosion • Water purification/detoxification: ecosystems can be a source of water impurities but can also help to filter out/decompose organic waste • Natural hazard protection, e.g. storms, floods, landslides • Bioremediation
Cultural services, i.e. nonmaterial benefits that people obtain through spiritual enrichment, cognitive development, recreation etc	• Spiritual and religious value: many religions attach spiritual and religious values to ecosystems • The inspiration for art, folklore, architecture, etc • Social relations: ecosystems affect the types of social relations that are established, e.g. fishing societies • Aesthetic values: many people find beauty in various aspects of ecosystems • Cultural heritage values: many societies place a high value on the maintenance of important landscapes or species • Recreation and ecotourism
Supporting services, necessary for the production of all other ecosystem services	Soil formation and retention • Nutrient cycling • Primary production • Water cycling • Production of atmospheric oxygen • Provision of habitat

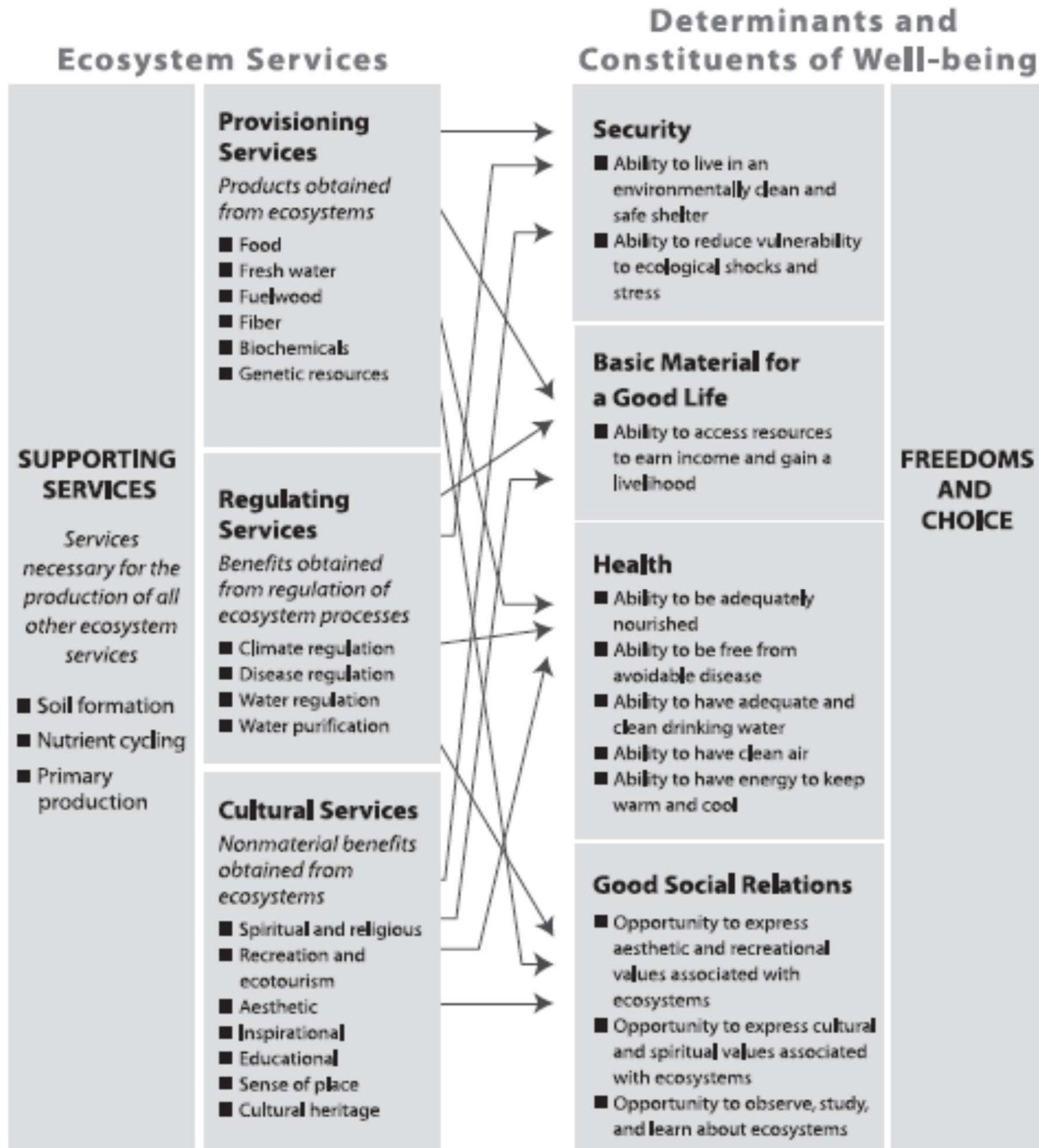

Figure 5: Ecosystem Services and their Links to Human Well-being (MEA, 2005)

Miombo woodlands support the livelihoods of 100 million people in the area or outside, relying on products from distinct and unique biome (Campbell and Frost 1996). '*Miombo*' is a term used to describe those central, southern and eastern African woodlands dominated by the genera *Brachystegia, Julbernardia* and *Isoberlinia*, three closely related genera from the legume family (Fabaceae, subfamily Caesalpinioideae). There are 21 species of *Brachystegia* in *miombo* woodland and three species of each of the related genera (Campbel, et al., 1996).

2.1.1 Miombo Woodlands as Carbon Sinks

Terrestrial carbon stock mapping is important for the successful implementation of climate change mitigation policies (Chave, et al., 2014). The ecosystems in general play a major role in global carbon budget and fluxes (Munishi, et al., 2010) changes in forest cover use and management produce sources and sinks of carbon dioxide (CO_2) that is exchanged with the atmosphere. Due to nutrient limitations, mature stands of Miombo woodlands show only a very limited annual increment in carbon storage (Campbell, 1996) and land use conversion, therefore, does not lead to a substantial additional loss of carbon sequestration capacity of the forests. For example, without land use and cover change, the miombo woodland is estimated to have carbon sink strength of about 0.39GtC year^{-1} (Lupala, 2015).

Average above-ground biomass in Miombo woodland in Zambia was estimated on average at about 20t/ha (Chidumayo, 1990). There are large variations among sites as a function of soil type, micro-climate, and past management, however, on average below-ground carbon content under Miombo was estimated at 78tc/ha and on agricultural fields at 42 to/ha (Walker & Desanker, 2004). In general, there has been a downward trend in the carbon stock of forest biomass in many miombo countries (Syampungani, et al., 2014) (Table 2).

Table 2: Trends in carbon stock in living forest biomass: selected Miombo countries 1990-2010

Country	Carbon stock in living forest biomass (10^6 Mg)				Annual changes (10^6 Mg yr^{-1})		
	1990	2000	2005	2010	1990-2000	2000-2005	2005-2010
Angola	4,573	4,479	4,432	4,385	-9	-9	-9
Malawi	173	159	151	144	-1	-1	-1
Mozambique	1,878	1,782	1,733	1,692	-10	-10	-8
Zimbabwe	697	594	543	492	-10	-10	-10
Zambia	2,579	2,497	2,497	2,416	-8	-8	-8

2.1.2 Miombo Woodlands and Livelihoods

A study conducted in 2002 to assess the contribution of miombo fruits to the livelihoods of the rural communities in the region has shown that 65–80% of rural households in the 'Chinyanja Triangle' i.e. Malawi, Mozambique and Zambia, lack access to food for as much as 3 or 4 months per year, and 26–50% of the respondents relied on indigenous fruits for sustenance during this critical period (Akinnefesi, et al., 2008). Firewood is the main source of energy in Zambia, in particular in the rural areas and is used for both cooking and heating (Hein et al., 2008). The maximum annual increase in wood biomass in miombo woodland is between 9–16 t/ha, and around half of this biomass can be harvested for construction purposes, e.g. timber, poles, tools, etc. (Campbel, et al., 1996).

Wild fruit is an important supplementary source of food and vitamins (Hein, 2009) in Tanzania. Miombo woodlands, 83 species of indigenous fruit trees have been recorded (Campbel, et al., 1996). Wild fruits are particularly important during times of food shortages (Hein, 2009).

Apiculture is a traditional occupation throughout the Miombo woodland zone. Productivity estimates from Tanzania indicate that 1km² can support 20 bee colonies producing 45kg of bees wax and 500kg of honey per year (Lowore, et al., 1995). Wet Miombo woodland has abundant and diverse mushroom populations. For example in Malawi, 60 species of edible fungi have been documented (Campbel, et al., 1996).

The roots, leaves, and bark of many different species are used in local health care. Some medicinal plant compounds are internationally marketed, such as quinine (Campbel, et al., 1996). Throughout the miombo, trees and woodland are important in the spiritual and cultural life of residents.

2.1.3 Ecosystem Services in Zambia

2.1.3.1 Forest Resources

The country's forest cover is estimated at 49.9 million hectares or 66% of the total land over of Zambia (GRZ, 2009). What the total growing stock (volume) across all land uses in Zambia is estimated at 2.9 billion m^3, with the majority of this volume, 2.1 billion m^3, held in semi-evergreen miombo dominated forests. The total biomass (i.e., above and below ground) is estimated at 5.6 billion tons, with additional 434 million tons of dead wood biomass, for total biomass estimated at 6 billion tons of this biomass, there are approximately 2.8 billion tons of carbon stored in the forests (GRZ, 2009). The potential for increased carbon sequestration from the terrestrial forests in Zambia is high due to high total growing stock of the forests and potential for reducing emission from forests (GRZ, 2009).

The honey industries engage an estimated 20,000 beekeepers country wide and an additional 6,000 honey hunters. It is estimated that annual domestic consumption of honey is 300 tons and 700 tons are used for beer brewing. About 50% of the domestic honey trade is consumed by the rural population, 36% is sold to traders, 8% is sold on the roadside, and 6% is traded in urban areas. The beekeepers have the potential of increasing honey export levels from around 600 metric tons annually to 2000 metric tons. Honey demand is forecasted to continue on an upward trend because of increasing consumer preference for organic products (GRZ, 2015).

In 2003, Zambia exported medicinal plants valued at an estimated US$ 4.4 million (Ng'andwe, et al., 2006). More than 80% of rural communities in Sub-Saharan Africa depend on the medicinal plant (Garrity, 2004). The forestry sector in Zambia contributes about 6.3% to the GDP (Turpie, et al., 2015).

2.1.3.2 Agricultural Resources

There is a general consensus throughout the literature (GRZ, 2009) (Hichaambwa & Jayne, 2014) that Zambia's large potential in agriculture remains unexploited, out of a total of 7.5 million hectares of land, 4.2 million hectares, (58%) for agricultural production for which 12% is suitable for arable production, with only an estimated 14% currently cultivated

2.1.3.3 Water Resources

Total water withdrawal from river systems was 1.737 km^3 in 2000, with agriculture use accounting for 1.320 km^3 (77 percent), or more than three-quarters of the total domestic water use claiming 0.286 km^3 and industries are taking 0.131 km^3 (GRZ, 2015). The total renewable water resources of Zambia amount to about 105 km^3 per year of which 80 km^3 are produced internally (World Bank, 2010) and while the renewable water resource per capital is estimated at 8,700m^3 per year, well above the average for sub-Sahara Africa (7,000m^3 per person per year) and global average (8,210m^3 per person per year). The main source of renewable water resources in Zambia is rainfall.

Zambia's endowment of water resources and topography provide significant hydropower resource potential, estimated at 6,000 megawatts (MW). The installed hydropower capacity represents only 27 percent of the country's hydropower potential and accounts for 99 percent of all electricity production in Zambia.

2.1.3.4 Fisheries Resources

Total production from capture fisheries is approximately 65,000 to 80,000 tons per annum, with an additional 5,000 metric tons estimated from the emerging aquaculture sector. Average per capita fish supply has declined from over 11 kg in the 1970s to approximately 6.5 kg in the 2000s (Musumali, et al., 2009). Contributions to rural economic growth and commerce provide significant economic opportunities for the poor. Although the fisheries sector provides income for over 300,000 people, such benefits are poorly quantified and often overlooked (Musumali, et al., 2009).

2.1.3.5 Wildlife Resources

According to the (GRZ, 2011), wildlife resources range from birds and reptiles to mammals. There are about 733 bird species (76 rare or endangered), 150 species of reptiles and 224 species of mammals (16 domesticated). Under the wildlife statutes, 25 species of mammals, 36 species of birds and four species of reptiles have international significance and are protected. In terms of area extent, wildlife protected areas occupy about 40 percent of the country's land surface area. This gives Zambia one of the largest wildlife estates in the sub-region. There is also scope for development of private wildlife estates, as there is the availability of land in communal (open) areas. Also, there are about 404 species of fish (204 endemics), and a few protected areas as fish breeding sites. There are also over 3,000 sites of outstanding natural and cultural heritage with a great diversity of attractions and unique selling features that have been identified in the Zambian landscape, including the Victoria Falls.

2.1.3.6 Wetlands and wetland resources

Dambo and floodplain wetlands are used for grazing animals in the dry season when upland vegetation is dry and with little nutritive value. They are also important for fishing, livestock-watering, hunting of small animals, a collection of thatching grass, and most importantly, for dry season vegetable growing (GRZ, 2015)

2.1.3.7 Mineral resources

According to the (GRZ, 2011), the mining sector has been a prime driver of economic development in Zambia for over 70 years, with exports of mineral products contributing about 70 percent of total foreign exchange earnings. Lately, mining has been generating between 6 percent and 9 percent of Gross Domestic Product (GDP) and contributes about 40,000 jobs to total formal sector employment. The role that the mining sector plays in national development is enhanced by its backward and forward linkages with other sectors. It provides critically needed inputs for agriculture and agro-chemicals, industrial manufacturing of a wide range of products such as ceramics, paint manufacture and essential raw materials for the building industry. In this regard, the mining sector' s contribution to wealth creation in the economy is far-reaching.

3 RESEARCH METHODOLOGY

This chapter describes the research materials and methods used to accomplish the study. A multiple-method was used in the research since the various information was needed to address the research questions. The chapter starts by describing the study area and follows it with the various methods used to gather data on the impact of deforestation on ecosystem services.

3.1 Study Area

This study site is part of the Kamfinsa sub-catchment of the Kafue river watershed in Kitwe District of the Copperbelt Province of Zambia (extending between 12.87 degrees and 13.00 degrees southern latitude and 28.30 degrees and 28.50 degrees eastern longitude at an elevation of 1,292 to 1,300 m above sea level). It has annual rainfall varying from 1064 to 1302mm per annum, and experiences three seasons namely: hot dry (September–November), rainy season (December–March) and the cold, dry season (April–August). The length of growing season (LGS) is 157 days starting in the first dekad of November and ceasing during the third dekad of March, and seasonal mean temperature of 21 degrees. According to Koeppen Classification, the climate is warm temperate with dry winters and hot summers and falling under dry sub-humid with aridity index of 0.64.

The vegetation of the study site is dominated by Miombo woodland, which characterizes most extensive tropical seasonal woodland and dry forest formations in Africa. Miombo is a vernacular word that has been adopted by ecologists to describe those woodlands dominated by trees of the genera Brachystegia, Julbernardia, and Isoberlinia Leguminosae, subfamily Caesalpinioideae (Campbell, 2007). The miombo region in Africa covers an estimated 2.4 million km^2. Above-ground biomass stocking densities of the Miombo vary from 20 m^3 per ha to as much as 150 m^3. Characteristically, miombo is found in areas which receive more than 700 mm mean annual rainfall and where soils tend to be nutrient-poor (Campbell *et al.* 1996).

Miombo woodlands cover substantial portions of southern Africa: Angola, Zimbabwe, Zambia, Malawi, Mozambique and Tanzania, and most of the southern part of the Democratic Republic of

Congo (DRC). It is dominated by a few species, mostly from the genera *Brachystegia, Julbernardia,* and *Isoberlinia.* Miombo is so-named after the Swahili word for a *Brachystegia* species. In the entire Miombo eco-region, Zambia has the highest diversity of trees and is the centre for endemism for Brachystegia tree species (Rodgers et al., 1996).

The soils in the study area are mainly the loamy sand (Acrisols). These soils are highly weathered and strongly leached and are thus infertile soils, characterized by weakly structured, loamy top soils, clayey sub soils. The soils are characterized by low base saturation due to leaching by excessive rainfall.

The Catchment starts from the source of Kamfinsa stream in Sakania area of Ndola rural district, bordering Democratic Republic of Congo (DRC). Kamfinsa stream drains the catchment with its five main tributaries, namely Chibwe, Mwambonalimo, Kafibale, Kamishishi and Kamatete streams (Figure 6). Kamfinsa stream flows into Kafue River at a confluence about 200 meters downstream from Kafue Bridge along the Ndola-Kitwe dual carriageway.

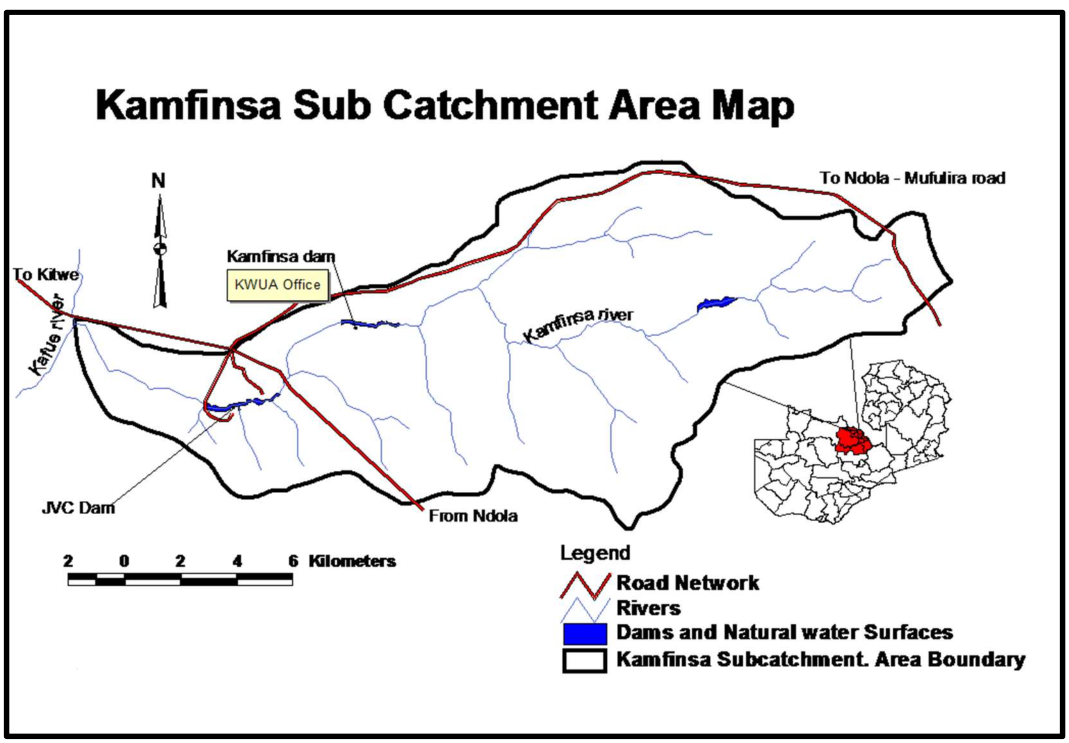

Figure 6: Kamfinsa Sub-catchment Area

The study utilized raster GIS environment to generate the RUSLE factors required to generate the annual soil erosion rates. RUSLE was developed as an equation of the main factors controlling soil erosion, namely climate; soil characteristics, topography and land cover management. Average annual soil loss was estimated by multiplying R, K, LS, C and P factors with use of Raster Calculator extension in SAGA (System for Automated Geoscientific Analyses) GIS software environment.

3.2 Research Methods

While quantifying the economic value of environmental services and disservices are useful for informed decision making, methodological difficulties remain an obstacle to the making of true comparisons (Rasul, 2009). A common approach to ecosystem service assessment is to use proxy variable particularly land cover, to represent ecosystem processes and provide maps of ecosystem services (Seppelt, et al., 2012). To allow true assessment of the impact of agriculture ecosystem services, it is necessary to capture key environmental services such as soil conservation, carbon sequestration, and biodiversity protection, along with marketable goods and services (Swinton, et al., 2007). Land use change, the main driving force of environmental change, is central to sustainable development (Jansena, et al., 2008).

This study focused on the following ecosystem services as proxies to assess the impact of deforestation on Ecosystem Services:

a. Provisioning Services: Genetic resources (forests),
b. Regulating Services: Climate regulation
c. Cultural Services: Educational values, Cultural heritage values
d. Supporting Services: Soil formation

3.2.1 Sampling Design

The stratified random sampling design was used in this study. The use of this sampling design has advantages, and the major ones are low cost, quick data collection, greater scope and accuracy (Cochran, 1977). If the population is not homogenous, it is possible to partition it into more or less homogenous groups called strata (Lemay & Marshall, 1990). The idea objective is to make the variation within the strata small and the variation among strata large resulting in smaller standard deviations for the estimates of the mean and hence better efficiency (Lemay & Marshall, 1990). Land use change results in different land cover/land use hence the need for stratification.

The sampling design was mainly through the use of GIS and remote sensing techniques to assess the whole sub-catchment area as well as forest inventory. This approach allowed for analyzing the proxy variables. The stratification for forest inventory was based on vegetation types due to land use change.

Circularly shaped sample plots were adopted because they are easy to use, they reduce or minimize edge effects in the samples and counting errors during an inventory of border trees. These edge effects are less on the circle plots than in square and rectangular plots (Krebs, 1989). The sample plot was divided into three-nest circular sampling plots of 1m, 10m, and 20m radius (Pearson, et al., 2005). The information that was recorded from each sample plot includes diameter at breast height (dbh), tree and shrub species names, and regeneration frequency.

According to (Pearson, et al., 2005), potential stratification options include:

a. Land use (for example, forest, plantation, agroforestry, grassland, cropland, irrigated cropland);
b. Vegetation species (if several);
c. Slope (for example, steep, flat);
d. Drainage (for example, flooded, dry);
e. Age of vegetation;
f. Proximity to settlement.

The number of sample plot in the stratified area was based on equation 1 and allocation of sample plots in each stratum was based on equation 2.

$$n = \frac{(\sum_{h-1}^{L} N_h * S_h)}{\frac{N^2 \ * E^2}{t^2} + \ (\sum_{h-1}^{L} N_h * S_h^2)}$$

Equation 1

Where:

E = *allowable error or the desired half-width of the confidence interval. Calculated by multiplying the mean carbon stock by the desired precision (that is, mean carbon stock x 0.1, for 10 per cent precision, or 0.2 for 20 per cent precision),*
t = *the sample statistic from the t-distribution for the 95 per cent confidence level. t is usually set at 2 as sample size is unknown at this stage,*
Nh = *number of sampling units for stratum h (= area of stratum in hectares or area of the plot in hectares),*
n = *number of sampling units in the population*
sh = *standard deviation of stratum h.*

$$n = n * \frac{N_h * S_h}{\sum_{h-1}^{L} N_h * S_h^2}$$

Equation 2

Where:

n = *the total number of plots,*
nh = *the number of plots in stratum h,*
N = *the number of sampling units in the population,*
Nh = *the number of sampling units in stratum h,*
s = *the standard deviation,*
sh = *the standard deviation in stratum h.*

3.2.2 Measurement and Data Collection

The sets of data used for analysis was based on variables generated using GIS and remote sensing techniques. Data was generated from Satellite Images (Landsat) on forest cover and digital elevation models (DEM). DEM is a digital model or Three Dimensional (3D) representation of a terrain surface, created from terrain elevation data. These were used to estimate the forest cover change, soil erosion and biodiversity change.

3.2.2.1 ***Estimation of Land cover Change***

The definition of Land cover considers the attributes of the earth's land surface captured in the distribution of vegetation, water, desert and ice and the immediate subsurface, including biota, soil, topography, surface and groundwater, and it also includes those structures created solely by human activities such as mine exposures and settlement (Lambin, et al., 2001). On the other hand, land use is the intended employment and management strategy placed on the land cover by human agents, or land managers to exploit the land cover and reflects human activities such as industrial zones, residential zones, agricultural fields, grazing, logging, and mining among many others (Zubair, 2006) .

Therefore, Land use change is defined to be any physical, biological or chemical change attributable to management, which may include conversion of grazing to cropping, change in fertilizer use, drainage improvements, installation and use of irrigation, plantations, building farm dams, pollution and land degradation, vegetation removal, changed fire regime, spread of weeds and exotic species, and conversion to non-agricultural uses (Quentin, et al., 2006).

Land use change was evaluated over a period of 20 years from data sets of satellite images acquired for the years 1990, 2000 and 2010. These satellite images were geo-referenced and maps developed and statistically analyzed for land cover/land use. This information is necessary for updating land cover maps and the management of natural resources (Xiaomei & Ronqing , 1999). Monitoring changes and time series analysis is quite difficult with traditional methods of the survey (Olorunfemi, 1983). Conventional ground methods of land use mapping are labour intensive, time-consuming and are done relatively infrequently and hence have become outdated with the passage of time, particularly in the rapid changing environment (Zubair, 2006).

Landsat images consisting of spectral bands covering the study period were used for landcover analysis. The bands were used for creating false colour composite images and subsequently used for the production of the land-cover maps. The false colour satellite images were later geometrically and projected to UTM Coordinate System, datum WGS 84, zone 35s.

Considering that vegetation was a key component in the land cover mapping, false colour composite images were generated using 3 bands of Landsat namely band 4 in near-infrared, band 5 in the mid-infrared and band 2 in the visible part of the spectrum. Later the false colour images were clipped using the Kamfinsa sub-catchment area map boundary to generate the thumbnail images for each of the years 1990, 2000 and 2010.

The classification of thumbnail Landsat satellite images covering the sub-catchment area was done using ArcGIS 10.1 - Spatial Analyst Extension. The classification method adopted was supervised classification using the Maximum Likelihood Classifier. The land cover maps were digitally generated based on 6 classes. Later the land cover maps which were in raster format were converted to vector format. Statistics for each land cover class were generated.

3.2.2.2 Inventory and Estimation of Biomass and Carbon Stock

Forest inventory is defined as the procedure for obtaining information on the quantity and quality of the woodland resources and other characteristics of the land on which the trees and shrubs are growing (Malimbwi, 1997). Direct causes of forest area changes vary within the country and estimation of biomass and carbon stock changes dependent on the way forest is cleared or converted to other land uses (Chidumayo, 2002). According to Chidumayo, (2002) the common cultivation practices in Zambia include (i) Chitemene shifting cultivation, (ii) block shifting cultivation, (iii) semi-permanent and commercial cultivation. The intensity of forest transformation under these land use types is different and so are their impacts on wood biomass and carbon stocks.

The study sites were stratified based on vegetation and forest inventory of both mature undisturbed woodlands and regrowth stands. In this case, the mature undisturbed woodland is one that was not severely affected by human activities, e.g. charcoal production, or clearing for agriculture purposes while the regrowth stand is one that was recovering from human disturbances.

The sample plot was divided into three-nest circular sampling plots of 1m, 10m, and 20m radius (Pearson, et al., 2005). The information that was recorded from each sample plot includes diameter at breast height (dbh), which is at 1.3 m above ground, tree and shrub species names, and regeneration frequency. The use of smaller plots in regrowth plots is due to the many species and high density of these plots which makes the use of larger fixed plots time consuming (Syampungani, et al., 2010). Tree volume was calculated using the following formula:

$$\pi * \mathrm{dbh}^2 * H * 0.74/4$$

Equation 1

Where H is tree height and 0.74 is a correction (form) factor.

However, (Endean, 1967), based on the Ndola Indigenous Sample Plots, noted that the best indicator of harvestable wood volume in indigenous forests was the Stand Basal Area (BA) and this was used to estimate the productivity of Miombo woodland. The two approaches for estimating the biomass of woody vegetation types are *the volume method* and *the direct biomass estimate method*. The volume method uses measured volume estimates that are then converted to biomass (tonnes/ha) using a variety of tools (Forestry, 2016). The direct estimates of the biomass method use biomass allometric equations, i.e., functions that relate oven-dry biomass per tree as a function of a single or a combination of tree dimensions (Brown, 1997).

Linear models can be based on logarithmically transformed data (log models) or square-rooted data and untransformed data (Chidumayo, 2016). Others use power models based on either dbh alone, or in combination with other predictors, such as tree height and crown diameter, or basal area, but a few use polynomial models. A linear equation was used in calculating the volume.

In case of estimating only harvestable wood in Miombo, a model by (Endean, 1967) was considered, which uses the basal area as a predictor variable. The basal area of a forest stand is found by adding the basal areas of all of the trees in an area and dividing by the area of land in which the trees are measured, and it is expressed as m^2/ha.

$$\text{Biomass (t/ha)} = 6.234\text{BA} - 15.54$$

Equation 2

Where biomass is aboveground wood mass (t/ha), and BA is a basal area at breast height (m^2/ha) (Chidumayo, 1990). Estimated mean annual increment (MAI) in aboveground wood biomass in regenerating forest as 2.0 ± 0.24 t ha^{-1}.

3.2.2.3 Estimation of Soil Erosion

Soil erosion is the wearing away of the land surface by physical forces such as rainfall, flowing water, wind, ice, temperature change, gravity or other natural or anthropogenic agents that abrade, detach and remove soil or geological material from one point on the earth's surface to be deposited elsewhere (Jones, 2007). Soil erosion is normally a natural process occurring over geological timescales; but where (and when) the natural rate has been significantly increased by anthropogenic activity accelerated soil erosion becomes a process of degradation and thus an identifiable threat to the soil.

Land cover changes have been used to predict soil erosion. Soil erosion is related to ecosystem services both in terms of water quality and future agricultural productivity (Dale, 2007). In addition, soil erosion rates are largely a function of the proposition of bare ground (Dale, 2007). Remote sensing has been used to assess changes in gulling demising and balance area (Keay-Bright & Bright, 2006). The Revised Universal Soil Loss Equation (RUSLE) was used to estimate annual soil loss from the study area (Prasannakumar, et al., 2012).

The soil loss from a unit area can be estimated by:

The soil loss equation is expressed as below:

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

Equation 3

Where A is the annual average soil loss - [Ton ha^{-1}year^{-1}],

R is the rainfall intensity factor - [MJ mm ha^{-1}h^{-1}year^{-1}],

K is the soil erodibility factor - [(ton h MJ^{-1} mm^{-1}],

LS is the topographical (slope-length) factor - [Dimensionless],

C is the land cover factor - [Dimensionless], and

P is the soil conservation or prevention practices factor - [Dimensionless]

The soil erosion model, RUSLE represents the links on how climate, soil, topography, and land use affect rill and interrill soil erosion caused by raindrop impact and surface runoff (Renard & Ferreira, 1997). The model has been extensively used to estimate soil erosion loss, to assess soil erosion risk, and to guide development and conservation plans in order to control erosion under different land-cover conditions, such as croplands, rangelands, and disturbed forest lands (Milward & Mersey, 1999). The process of generating the RUSLE variables is illustrated in Fig. 7.

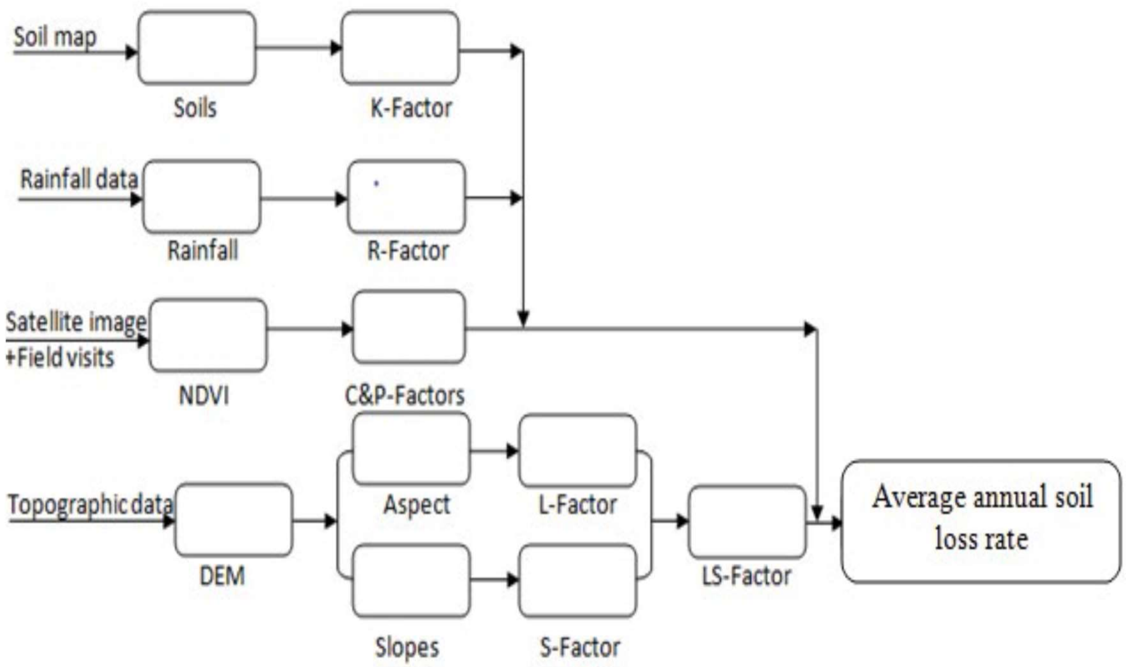

Figure 7: Methodology to generate RUSLE factors

3.2.2.3.1 Soil Erodibility (K) Factor

The Soil erodibility Factor (K) values represent the susceptibility of soil to erosion. The soil erodibility factor K is a quantitative value which is experimentally determined to take into consideration the soil texture, soil structure, the organic matter content and the permeability (Wischmeier, et al., 1971). Determination of the soil erodibility involves assigning values that correspond to the soil types contained within the research area (Torri, et al., 1997)..

Based on (Wischmeier, et al., 1971), K was calculated

$$100K = 2.1 * 10^{-4} * M^{1.14} * (12 - OM + 3.25\ (S - 2) + 2.5\ (P - 3)$$

Equation 4

Where

$$M = [\%\text{very fine sand} + \%\text{ silt}]\ \text{X}\ [100 - \%\text{ clay}]$$

Equation 5

OM is Percentage of organic matter;

S is the Code according to the soil structure (very fine granular = 1, fine granular = 2, coarse granular = 3, lattice or massive = 4), and

P is the Code according to the permeability/drainage class (fast = 1, fast to moderately fast = 2, moderately fast= 3, moderately fast to slow = 4, slow = 5, very slow = 6).

This study utilized the default, FAO, Harmonised World Soil Database of 2012 and (GRZ, 2009) to calculate the soil loss on-site. The soils at Kamfinsa sub-catchment area are predominantly Acrisols. The K-factor under the RUSLE was calculated from (top soil 0-30cm); Sand 60%, Silt 14%, clay 26% and bulky density of 1.41%.

The soil erodibility was calculated using the soil properties extracted from the Harmonised World Soil Database (2012) using equation 4 (Wischmeier, et al., 1971). This equation was settled upon due to the availability of data on soil structure, organic matter, and permeability.

However, the HWSD database, Version 1.2 (2012) does not provide organic matter content directly, but in terms of a ratio of organic carbon in gram/kilogram soil. This ratio was converted to a percentage, and then multiplied by a factor of 1.72414 to convert it to organic matter as shown in equation 6 below.

$$Organic\ Matter\ (\%) = Organic\ Carbon\ (\%)\ X\ 1.72414$$

Equation 6

According to this conversion factor, it was assumed that organic matter contains 58% organic carbon (Wayne, et al., 1992).

3.2.2.3.2 Rainfall-Runoff Erosivity (R) Factor

Rainfall and runoff play an important role in the process of soil erosion, which is usually expressed as the *R* factor. To calculate the *R* factor, long-term precipitation data with high temporal resolution were used, this data is typically available or only rarely available at standard meteorological stations. Consequently, long-term average R-values are often correlated with more readily available annual rainfall or the modified Fournier's index (Arnoldous, 1980).

The need for very specific data often makes the calculation of R according to its definition impossible and has resulted in derivation of simplified methods. These are based either on the total annual precipitation P or Fournier Index F (Arnoldous, 1980).

Different authors have proposed different ways to retrieve R from P or F. To determine the suitable equations that could be used at a given catchment, equations given in Table 3 were evaluated based on the R factors provided in Table (Kassam, et al., 1992).

Kamfinsa sub-catchment area being a small catchment, the equation in Table 3, case VII was used for calculation.

Table 3: Reference Equation for Estimating R-factor

Case	Reference	R and P or F Relationship
Case I	Arnoldous – linear, 1980	$R = (4.17F-152)/17.02$
Case II	Arnoldous (1980)	$R = 4.17F - 152$
Case III	YU & Rosewell, 1996	$R = 3.82\, F^{1.41}$
Case IV	Arnoldus – Exponential, 1977	$R = 0.302\, F^{1.93}$
Case V	Renald & Freimun – F, 1994	$R = 0.739F^{1.847}$
Case VI	Renald & Freimun – P, 1994	$R = 0.0483P^{1.61}$
CaseVII	Roose in Morgan and Davidson (1991)	$R = P \times 0.5$

Where

R = rainfall erosivity factor (MJ/ha.mm/h)
F = Founier index
P: Mean Annual Precipitation (mm)

The rainfall data used to calculate R under this study was collected from the Meteorological Department as mean monthly rainfall data for a 10 years period (1996 to 2006) from the nearest weather station (Ndola).

3.2.2.3.3 Soil Cover (C) Factor

The Cover-factor represents how management exposes soil to erosive agents. High values of C-factor imply high erosion potential. The value of *C* depends on vegetation type, stage of growth and cover percentage. Removal of vegetation can greatly increase the runoff and soil erosion, particularly in mountainous areas (Gurevitch, et al., 2002). The NDVI, a spectral ratio between near infrared and infrared reflectance, extracted from satellite imagery can be used to calculate the *C* values, bearing in mind that the NDVI is highly correlated with vegetative cover and biomass (Zhang, 2002).

The effect of vegetation cover on soil erosion is mainly assessed by three different ways. The first one is a direct application of the C-factor on the RUSLE based on; prior land use (PLU), canopy cover (CC), surface cover (SC), the surface roughness (SR), and soil moisture (SM) sub- factors (Renard & Ferreira, 1997). The second one is to assign a C-factor according to a qualitative ranking of vegetation types (Wischmeier & Smith, 1978). The third method is to calculate the C-factor from the Normalized Difference Vegetation Index (NDVI), defined as the near-infrared reflection minus the red reflection divided by the sum of the two (Wu, et al., 2004). Satellite-derived NDVI has been particularly useful for assessing vegetation (Viedma, et al., 1997).

The traditional field methods are not feasible in the large-scale estimation of RUSLE factors (Curran & Williamson, 1986). Remote sensing data becomes key since satellite imagery provides a spatially and periodic, comprehensive view of land vegetation cover (Chen, et al., 2005). For generating canopy cover values, normalized difference vegetation index is computed using near-infrared (NIR) and red band (R), and using the conventional equation (Ioannis, et al., 2009) .

$$\mathrm{NDVI} = \frac{\mathrm{NIR} - \mathrm{IR}}{\mathrm{NIR} + \mathrm{IR}}$$

Equation 7

Where

NIR**:** The reflection of the near infrared portion of the electromagnetic spectrum and

IR**:** The reflection in the upper visible spectrum

The NDVI values lie in the range [-1, +1], but (Ioannis, 2009) indicates that vegetation traces are detected in values bigger than +0.18

Since the original *C*-factor of USLE ranges from 0 (full cover) to 1 (bare land) and the *NDVI* values range from 1 (full cover) to 0 (bare land), the calculated *NDVI* values are inversed. The *C*-factor map was produced using the following exponential equation (Van der Kniff, et al., 2000).

$$C = -\alpha \frac{NDVI}{\beta - NDVI}$$

Equation 8

Where

α, β: parameters determining the shape of the NDVI-C curve. An α-value of 2 and a β-value of 1 seem to give reasonable results (Ioannis, et al., 2009). (Van der Kniff, et al., 2000) found that this scaling approach gave better results than assuming a linear relationship. The C-factor has greater uncertainty for the lower range NDVI values due to non-photosynthetic vegetation (NPV) that is not measured by the NDVI as well as soil reflective properties.

3.2.2.3.4 Slope and Slope Length (LS) Factors

The slope and slope length factors (*S* and *L*, respectively) account for the effect of topography on soil erosion. The LS factor can be estimated through field measurement or from a digital elevation model (DEM). The slope length factor L is defined as the distance from the source of runoff to the point where deposition begins, or runoff becomes focused into a defined channel. The interaction of angle and length of the slope has an effect on the magnitude of erosion. As a result of this interaction, the effect of slope length and degree of slope should always be considered together (Morgan & Davidson, 1991). With the incorporation of Digital Elevation Models (DEM) into GIS, the slope gradient (S) and slope length (L) may be determined accurately and combined to form a single factor known as the topographic factor LS. The precision with which it can be estimated depends on the resolution of the digital elevation model (DEM).

In the process of determination of this factor, the slope length (L) and the slope(S) are first determined and then used together in coming up with the LS-factor.

The equation below was used in combining the two factors (S and L) to come up with the LS factor (Morgan & Davidson, 1991).

$$LS = \sqrt{\frac{1}{22}}(0.065 + 0.045 * \ +0.0065 * S^2$$

Equation 9

Where:

L = Slope Length
S = Percent Slope

3.2.2.3.5 Support Practices (P) Factor.

The support practice factor (*P*-factor) represent the soil-loss ratio with a specific support practice to the corresponding soil loss with up and down the slope (Renard & Ferreira, 1997). The P-factor ranges from 0 to 1, in which the highest value is assigned the areas with no conservation practices. Lower values of P indicate more effective conservation practices (Prasannakumar, et al., 2012).

In this study, the P-factor was given a uniform weight of 1.0 by the use of the shapefile bounding the catchment area. This is because there is no significant support practice in the area which is mainly associated with small-scale farming.

3.2.2.4 Assessment of Biodiversity loss and Change

Land use change is an important driver of biodiversity change as natural areas are converted to agriculture or urban areas. It is recognized that there is a relationship between species richness and primary productivity (Waide, et al., 1999)It is possible to use remote sensing to monitor biodiversity (Barber, 1983).

It is recognized that NDVI correlates directly with vegetation productivity as a result, there are numerous possible applications of this index for ecological purposes (Pettorelli, 2005).

The Normalized Difference Vegetation Index (NDVI) and Enhanced Vegetation Index (EVI) have been used to monitor global vegetation conditions as well as in modelling climatic, biogeochemical and hydrologic processes, or land surface biophysical processes, such as primary production or land cover conversion (Justice, et al., 2002). Using of NDVI in ecological studies, especially on environmental change in an ecosystem context, is well documented (Gao, et al., 2012) (Gould, 2000) (Pettorelli, 2005). These vegetation indices can effectively be used to represent environmental change in a wide range of contexts, including habitat destruction, degradation and fragmentation (Holm, et al., 2003), and can be used in correlative studies to identify declines in population abundances (Carey, et al., 2001) or species richness (Bar-Massada, et al., 2012).

Originally, the NDVI was used to generate maps, including the pioneering mapping of vegetation distribution and productivity in Africa (Tucker, 1985). The ecological relevance of such maps is multiple: the NDVI enables the differentiation of ecosystem functional types or biozones (Soriano & Paruelo, 1992), (Paruela, 2001), quantify the annual net primary productivity (ANPP) at various scales worldwide and to differentiate land cover at the continental (Kerr & Ostrovsky, 2003) and global (Nemani & Running, 1997) scales. By using the NDVI, it is possible to differentiate savannah, dense forest, non-forest and agricultural fields in Africa (Achard & Blasco, 1990) and in Asia (Achard & Estreguil, 1995). In addition, the phenological characteristics help to determine evergreen forest versus seasonal forest types (Van Wagtendonk, 2003) or trees versus shrubs (Senay & Elliot, 2002).

In undertaking the study, the NDVI for 1990, 2000 and 2010 were stacked using ArcGIS and generated an image in order to assess the change and potential impacts on biodiversity. The image was used to assess the change in the vigor of plants. The vigour was estimated through analysis of satellite images to indicate “green-ness.” Vegetation canopy characteristics such as biomass and percentage of vegetation cover is representative of plant photosynthetic efficiency, and fluctuations due to changes in meteorological and environmental parameters (FAO, 2005)

3.2.2.5 Estimating the value of forests lost due to Deforestation

The valuation method used in assessing the cost of deforestation on the ecosystem services was the Replacement Cost method. This method involves estimating the expense of replacing an ecosystem service with a man-made product, infrastructure or technology (Noel & Soussan, 2009). In order to calculate the value of forests deforested, the economic value of carbon was used in this study. Many recent forest valuation studies have used social cost of carbon emissions or the market value of carbon to estimate the value of forests (Turpie, et al., 2015).

The method was used to assess the following:

a. The costs of replacing the lost soil nutrients calculated from the soil loss equation in equation 3 above and used the current costs of fertilizers.
b. The cost of forests lost due to deforestation, calculated by assessing the biomass loss (carbon) of the areas cleared of the forest using the current price of carbon on the carbon market.

3.2.2.5.1 Soil Nutrient Loss (Replacement Cost Method)

There are three common and intuitive methods of estimating the on-site cost of soil erosion are the change in productivity method, the defensive expenditure method and the replacement method (Cho & Rapera, 2010). The replacement cost method assumes that the on-site cost of soil erosion due to decline in soil fertility may be estimated by monetary cost of restoring or replacing the lost soil fertility. However, according to (Barbier & Bishop, 1995), this method might arrive at an over estimate because lost fertility is usually estimated by comparing the soil fertility levels of soils from one farm with observed erosion and those from a control farm that is assumed to have zero erosion during the same period.

In assessing Soil erosion, Remote Sensing (RS) and Geographic Information System (GIS) technologies were used for erosion risk mapping due to their capabilities of handling and manipulating both attribute and spatial data. The inputs to this model were spatial, and they include the LS (topographic factor), K(soil erodibility factor), R(rainfall erosivity factor), C(cover management factor) and P(support practices factor). The topographical factor was determined from the 30m raster DEM of the catchment area; the K-factor obtained from the soil map (default) of the area and the R- factor obtained from the long-term Meteorological Department and FAO precipitation data. Based on the DEM, the average elevation of the area is 1267m (Fig 12) above sea level. The C-factor was determined from the Landsat ETM+ image covering the catchment area while the P-factor was assigned by the GIS software.

The Annual soil loss per hectare per year (t ha^{-1} $year^{-1}$) was used to calculate the on-site soil erosion cost (Annex 1).

4 RESULTS AND DISCUSSION

4.1 Land Cover Change

The land cover maps revealed the changes in 1990, 2000 and 2010 (Table 4) and (Figure 8). The indigenous forest reduced from 13,430.50 ha in 1990 to 7,081.20 ha in 2000 and only 2,904.70 ha in 2010. A total of 10,525.80 ha was lost in 20 years. This provides an average of 576.29 ha per year. On the other hand agriculture land increased from 2,272 ha in 1990 to 8,333 ha in 2000 and 12,251 ha in 2010. On average agriculture land increased by 9,979 ha in 20 years showing an annual increment of 498.95 ha. Grass land reduced in size as well from 1,009 ha in 1990 to 711 ha in 2010. The water body also reduced from 11.5 ha in 1990 to 9.3 ha in 2010. The plantations remain the same during this period on 8,035 ha. Results on land cover for the Kamfinsa area are presented in Table 3, Figure 8 and Figure 9.

Deforestation resulted in the loss of 10,525.80 ha (78.37%) in 20 years. This is an average of 576.29ha per year. This is higher than the average rate of deforestation of the country, which was 300,000 ha per annum about 0.60% (GRZ, 2009), The loss of the forest contributed to emission of 43.73t Carbon above ground biomass. This is at the cost of about US$243.60. However, forests are not just carbon, as other ecosystem services were lost in the process. The main cause of deforestation in the Kamfinsa sub-catchment area was agriculture activities caused by new settlements due to job loss from the mines on the Copperbelt (Shitima, 2005).

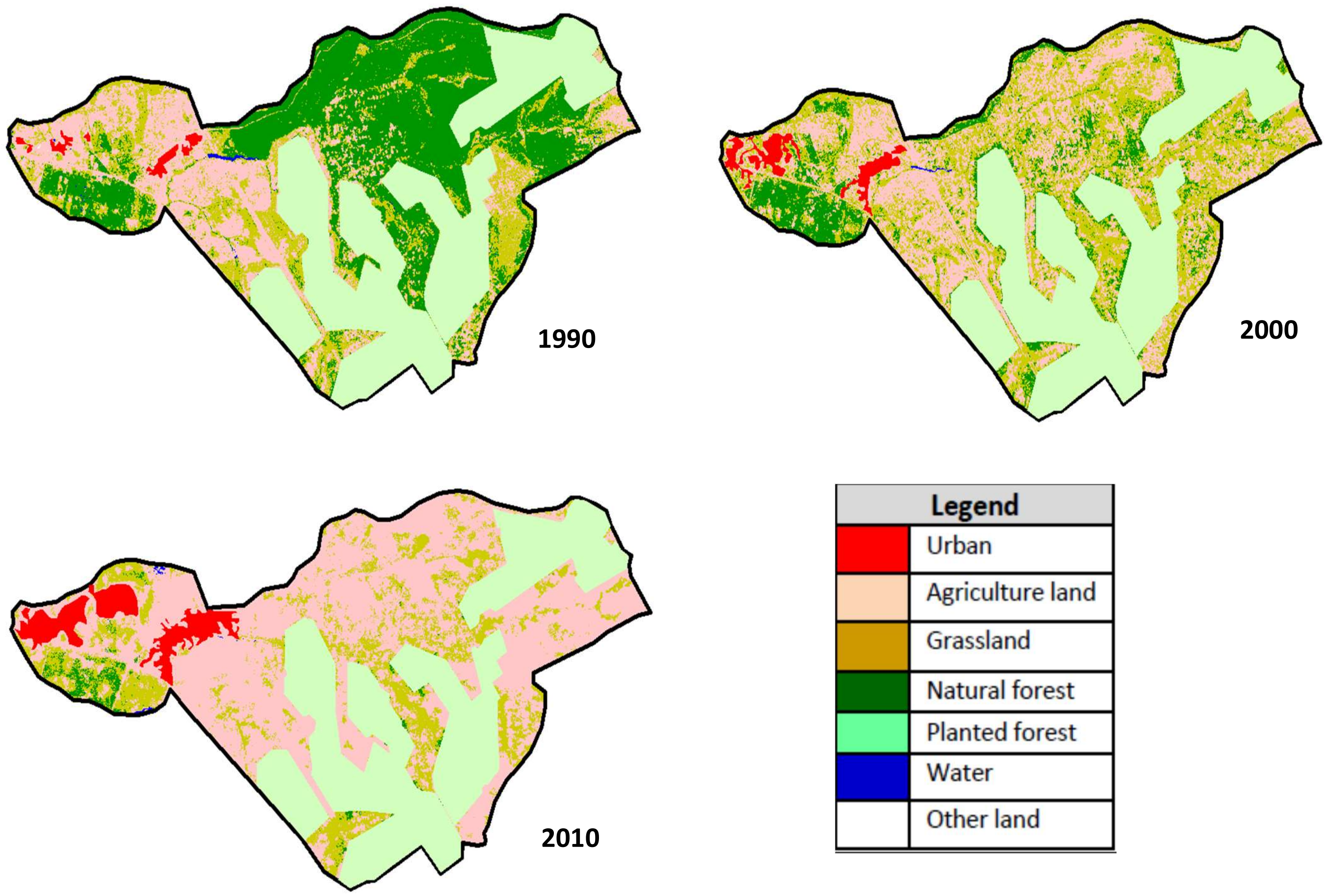

Figure 8: Land Cover Change Maps 1990, 2000 and 2010

Table 4: Landcover Change during the year 1990, 2000 and 2010

Classes	**1990**	**2000**	**2010**
	(ha)	**(ha)**	**(ha)**
Urban settlement	183.0	500.0 (173%)	1,030.0 (462%)
Agriculture land	2,272.0	8,333.0 (266%)	12,251.0 (439%)
Grasslands	1,009.0	983.0 (-3%)	711.0 (-30%)
Indigenous Forest	13,430.5	7,081.2 (-47%)	2,904.7 (-78%)
Plantation	8,035.0	8,035.0 (0%)	8,035.0 (0%)
Surface Water body	11.5	8.8 (-23%)	9.3 (19%)
TOTAL	**24,941.0**	**24,941.0**	**24,941.0**

In the Kamfinsa sub-catchment area, the plantations were the only type of forest that was persistent over the 20 year period. However, according to (Shitima, 2005), the forests in Mwekera provided other wood and non-wood forest products like charcoal, while (Kalaba, et al., 2013) indicated that the forests provided building poles, mushrooms, wild fruits and roots which local communities traded to support their livelihoods. In addition, (Kalaba, et al., 2013) indicated that the vegetation composition of regrowth sites suggested that pre-disturbance land use affected the vegetation composition during recovery. This meant that the loss of the forests would affect future biodiversity composition at the species level. This would, in the end, affect the livelihoods of local communities since the previously available biodiversity resources may no longer be available.

According to (Hein, 2009) the miombo woodlands provides overall revenues from the harvest of fruit, mushrooms, and honey of about US$5–15/ha/year, though large variations occur as a function of ecological characteristics (rainfall, flora composition) and economic variables (in particular distance to roads and cities). This helps local communities to meet some of their needs. (Kalaba & Quinn, 2013) Indicate that the major economic activities of the local communities in Kamfinsa sub-catchment area included agriculture (87%), charcoal production (33%), selling of mushrooms 27%, indigenous fruits (26%) and other 34% (beekeeping, beer brewing, and casual labour). Therefore, deforestation has a greater impact on the ecosystem services provided by the forests in Kamfinsa sub-catchment area.

4.2 Accuracy Assessment

The accuracy of the land cover classification from supervised and unsupervised techniques was evaluated and presented as an error or confusion in the form of matrix table (Hasmadi, et al., 2009). Accuracy for the Land cover maps was assessed using the confusion matrix (Tab. 5). The aim of accuracy assessment was to quantitatively assess how effectively the pixels were sampled into the correct land cover classes (Rwanga & Ndambuki, 2017). Various accuracy assessment and errors were calculated. Error matrix, which is also called confusion matrix, use the classified and reference data to compare and set up a matrix and it is a very effective way to calculate map accuracy (Ji & Niu, 2014).

The indexes, such as the overall accuracy, producer's accuracy, and user's accuracy, can be generated from the matrix (Ji & Niu, 2014). The Overall accuracy is an indication of the proportion of the reference sites, which were mapped correctly. On the other hand, errors of omission refer to the reference sites that were left out (or omitted) from the correct class in the classified map. The error of commission indicate the classified sites that were incorrectly classified

The producer accuracy shows the probability that a certain land cover of an area on the ground is classified as such on a map. In addition, User accuracy is the reliability that a class on a map will be present on the ground. Further, Kappa Coefficient evaluates how well the classification performed as compared to just randomly assigning values.

In assessing the accuracy of the maps generated, a confusion matrix was used for the period under review. The result of 2010 (Table 5) land cover map returned an overall accuracy of 86%, while 2000 was 81 % and that of 1990 was 79 %. All the land cover classes overall Kappa coefficient of 0.82.

Table 5: Accuracy Assessment for 2010 Land cover Map

2010 Land cover Acc. Assessment		Reference Data						Row Total	User Accuracy	The error of Commission. %
		Forest	Grassland	Cropland	Wetland	Settlements	other land			
Classified Data	Forest	161	9	7	1	0	0	178	90.4%	9.6
	Grassland	5	57	5	3	0	0	70	81.4%	18.6
	Cropland	6	7	43	0	1	1	58	74.1%	25.9
	Wetland	1	1	1	38	0	0	41	92.7%	7.3
	Settlements	0	0	1	0	8	0	9	88.9%	11
	Other-land	0	0	0	0	0	2	2	100%	0
Column Total		173	74	57	42	9	3	358	-	
Producer Accuracy		93.1%	77%	75.4%	90.5%	88.9%	66.7%	-	Overall accuracy (86.3%)	
Error of Omission. %		7	23	25	11.0	9.5	33			

4.3 Biomass loss

According to (GRZ, 2009), the average biomass stocking density in Mwekera-Kamfinsa indigenous forests was 81m^3/ha. This is equivalent to 38.07 tonnes of Carbon per hectare from above ground biomass (47% of total above ground biomass was considered as carbon based on Inter governmental Panel on Climate Change best practices,1996) and below ground biomass of 10.66 tonnes of Carbon per ha (based on IPCC that below/above ground ratio is 0.28 for tropical dry forests with above ground biomass of 20 tonnes/ha). The above ground biomass results compare well with those generated by (Kalaba, et al., 2013) of 39.9 tons per hectare.

The growing stock of the forest in the Kamfinsa Sub-catchment area was 1,087,868 m^3 in 1990, which reduced to 573,577 m^3 in 2000 and only 234,281 m^3 in 2010 (Table 6).

Table 6: Cost of deforestation in Kamfinsa Sub-catchment

Mean diameter (cm)	Average Volume/ha	Carbon stock/ha (AG and BG)	Deforestation rate (ha/year)	Price of Carbon/ton	Cost
16.57±0.21	81m^3	48.74 tonnes	576.29	US$5	US$243.65

Note: US$1 = ZKW 7

It will be noted that from the land-use change between 1990 and 2010, the Kamfinsa sub-catchment lost an average of 576.29 ha per year. Considering that each hectare has an average of 38.07 tonnes of Carbon above ground (AG) and 10.66 tonnes of carbon per hectare below ground (BG) (48.73 tonnes per hectare of total above and below carbon), this gives a total loss of about 21,938.22 tons of carbon above ground and 6,143.25 tonnes per hectare per year of below ground Carbon annually. Therefore, a total of 28,081.47 tonnes is emitted from the study area annually.

The project based price of carbon in 2015 at international level was about US$5 per ton of carbon. The cost to the ecosystem by losing a hectare of forest is, therefore, US$243.65. Considering that 576.29 ha is lost annually at 38.07 tonnes of Carbon per hectare above ground biomass and 10.66 tonnes below ground biomass. The total cost is US$196,570.29. This is equivalent to ZMK 1,375,992.03 (US$1=7 ZMK).

4.4 Soil Erosion

The mean values of the LS, K, R, C and P factors were 0.44 t h MJ^{-1} mm^{-1}; 0.01; 542 MJ mm ha^{-1} h^{-1} $year^{-1}$, 0.66 (Table 7) respectively and the support practices assumed and hence given a uniform weight of 1. After all the RUSLE factors were determined through GIS, they were combined in the GIS environment to come up with the soil erosion risk map of the area indicating different levels of soil erosion risk.

The results of soil erosion in the Kamfinsa Sub-catchment area are presented in Table 1 and Figure 10 and Figure 11:

Table 7: Table showing the values of R, K, LS, C and P of the RUSLE

A	R	K	LS	C	P
1.59t ha^{-1}	542 MJ mm ha^{-1}ha^{-1}year^{-1}	0.01th MJ^{-1}mm^{-1}	0.44	0.66	1

The resulting map for the study (Fig.10) shows Soil loss values ranged between 0 and 16.78 t ha^{-1} year^{-1} at the pixel level, with a mean value of 1.59 t ha^{-1} year^{-1} (Table 8). The Standard deviation was 1.37. As observed in Figure 10 and Table 9, the average high annual soil loss occurred in areas with high altitude and along the streams with steep banks and cultivation as these are seen to be targeted for cultivation. According to (Turpie, et al., 2015), it was estimated that average annual soil erosion on the Copperbelt area was 2t ha^{-1}.

Table 8: Soil Loss statistics for Kamfinsa Sub-catchment

Parameter	Maximum	Minimum	Mean	Standard deviation
Soil Loss (t ha^{-1} year$^{-1)}$	16.78	0	1.59	1.37

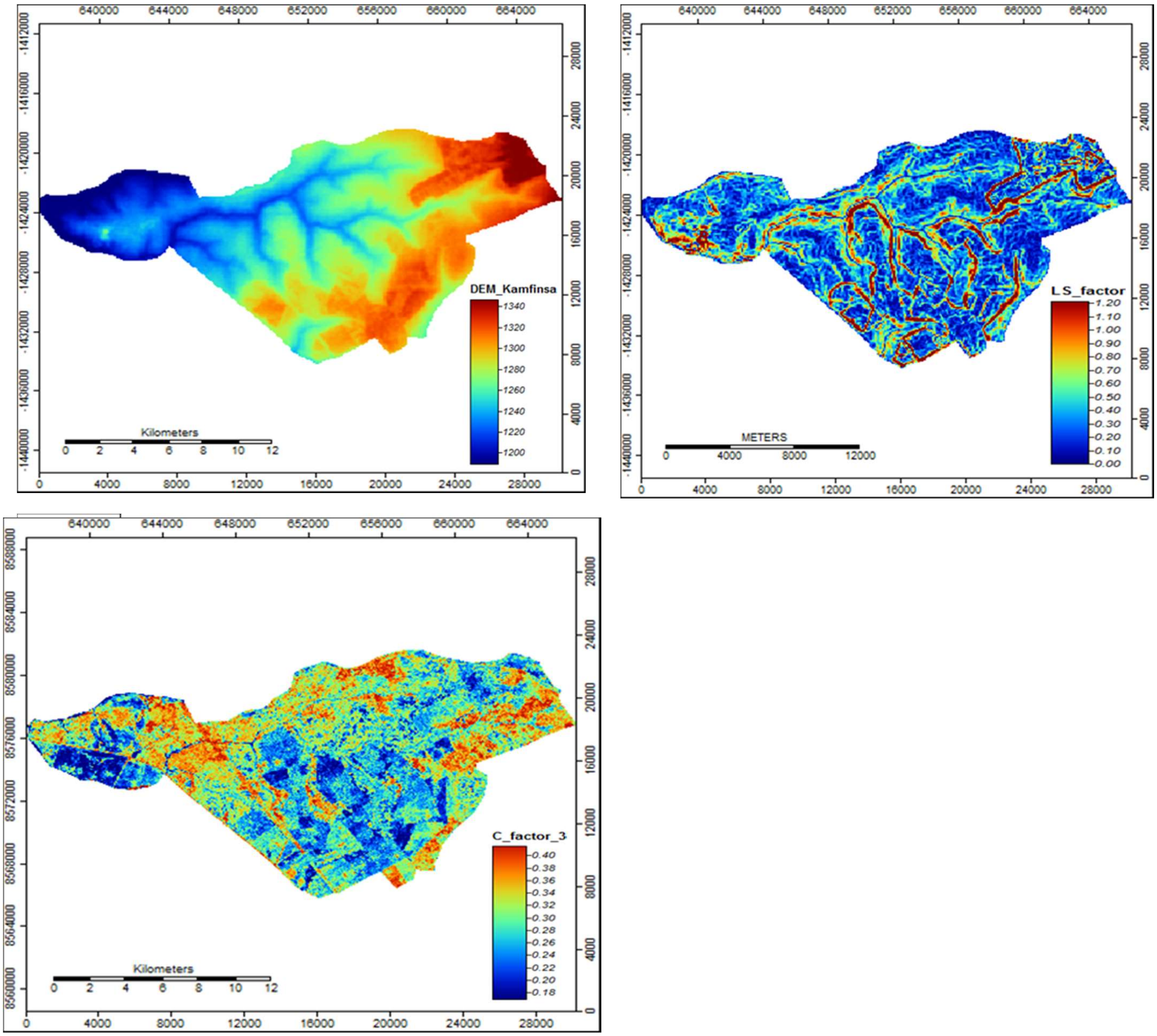

Figure 9: Kamfinsa Sun-catchment DEM, C-factor and LS-factor for soil loss estimation

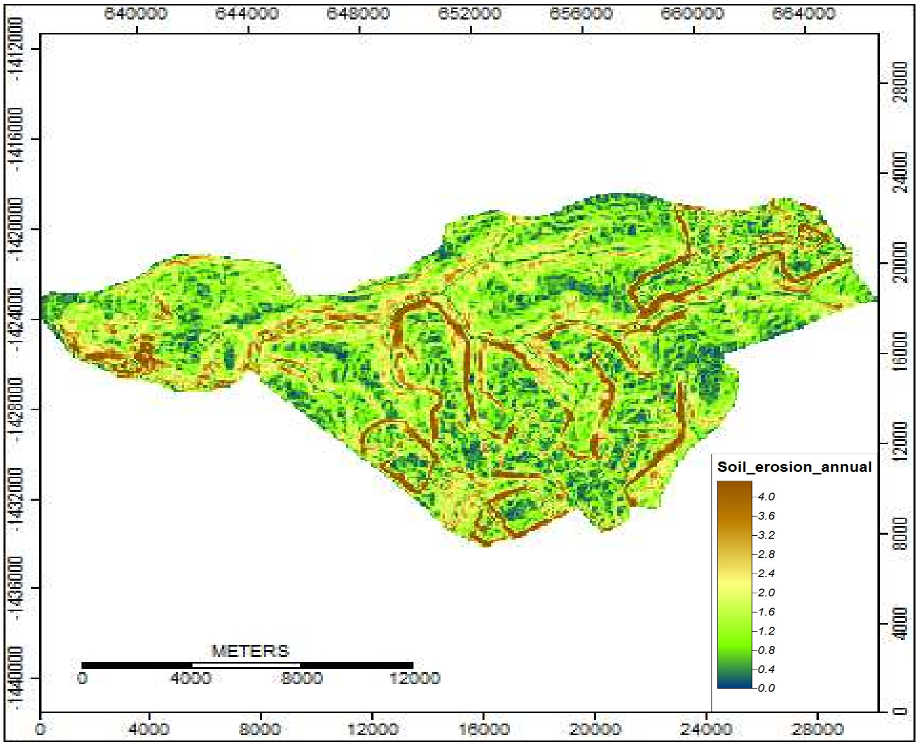

Figure 10: Annual Soil Loss

Table 9: Soil Erosion rates per area

Erosion Risk		Area (ha)	Soil Erosion (t ha^{-1} $year^{-1}$)
Severe		58.30	4
High		174.9	2.4
medium		21,803.60	1.20
low		2,904.70	0.40

4.5 On-site Soil Erosion Cost

In Table 10 a summary of the soil erosion related variables are presented, while the full calculations are in Annex 1. Based on the calculations, the amount of nutrients in 1 ton of undisturbed top soil was found to be 0.396 Kg. In terms of the cost of fertilizer containing 1 kg of the nutrient, it was found to be ZMW 125. The number of bags of fertilizer containing 1 kg of nutrient was 0.5 bags.

The on-site cost due to nutrient loss from 1ton of eroded soil is equal to the monetary value of specific inorganic fertilizers needed to replace the N, P and K contained in that ton of eroded soil.

Table 10 shows that replacement cost of nutrients in one ton of eroded topsoil in terms of inorganic fertilizers is ZMW 251.75 (the US $35.96). This is only for N, P, and K and does not include other nutrients and materials that are naturally found in topsoil. This amount, therefore, under-estimates the true cost of soil nutrient losses from one ton of eroded topsoil.

The study found that the cost of 1 ton of soil lost was ZMW 251.75 (the US $35.96). The soil erosion risk assessment found that 1.59 ton of soil was lost per hectare every year (Figure 10), and this means that the total cost per hectare is ZMW 400.28 (US$ 57.18).

Table 10: Estimated totals of on-site nutrients loss and rehabilitation costs

Quantities and Costs	Soil Nutrient and inorganic Fertilizers (Urea)			
	N	P	K	TOTAL
Amount of Nutrient in 1 ton of undisturbed top soil (kg)	0.068	0.312	0.016	**0.396**
Bags of fertilizer containing 1 Kg of the nutrient (bags)	0.2	0.1	0.2	**0.5**
Cost of fertilizer containing 1 Kg of the nutrient (ZMK)	50	25	50	**125**
Cost of fertilizer to replace the nutrient loss in 1 ton of eroded topsoil (ZKW)	170.00	78.00	3.75	**251.75**

1US$=7ZMK

The study observed that cost of 1 ton of soil lost was ZMK 251.75 (the US $35.96). The soil erosion risk assessment was estimated at 1.59 ton as soil loss per hectare per year (Figure 14), and this meant that the total cost per hectare is ZMK 400.28 (US$ 57.18). This value is only for N, P and K and does not include other nutrients and materials that are naturally found in topsoil. The amount, therefore, was an under-estimation of the true cost of soil nutrient losses from one ton of eroded topsoil. According (Amegashie, et al., 2012) and (Quamsah, et al., 2000), while it was useful to know the magnitude of soil nutrient losses, their on-site costs were equally important.

The increased area under cultivation from 2,272 ha in 1990 to 12,251 ha in 2010 increased the exposure of the land to soil erosion. It was also observed that in the cultivated field, there were no specific conservation measures that were being practiced. The area converted to agriculture was mainly from the indigenous forests.

It is clear from the study that opening up forests to facilitate crops resulted in an increased soil erosion through increased exposure to agents of erosion, mainly rainfall. It was important to improve land use planning in order to reduce the rate of soil erosion. In the time of climate change, it has been recognized according to (Henao & Baanante, 2006) that population growth and migration associated with drought, food shortages, and land overuse have accelerated degradation of agricultural land.

According to (Shawa, 2012), Mwekera dam (within the Kamfinsa sub-catchment area) was affected due to soil erosion resulting in siltation in the stream and the dam is the only water source for the National Aquaculture Research Development Centre (NARDC) and the Zambia Forestry College (ZFC). (Shawa, 2012) further observed that the dam was shallower due to siltation affecting the flow and fish production, especially in the dry season. The Mwekera community also complained of a reduction in fish catch due to shallow waters in the dam. In addition, the famous Mwekera falls were no longer flowing in the dry season as it used to do in the previous years.

Deforestation resulted in the loss of various ecosystem services including increasing stream flow hence siltation (Mumeka, 1986). In addition, according to (Shitima, 2005), the forests in Mwekera provided other wood and non-wood forest products like charcoal, while (Kalaba & Quinn, 2013) indicated that the forests provided building poles, mushrooms, wild fruits and roots which local communities traded to support their livelihoods. Further, (Kalaba, et al., 2013) indicated that the vegetation composition of regrowth sites in Mwekera suggested that pre-disturbance land use affected the vegetation composition during recovery.

4.6 Change in Biodiversity

The NDVI for 1990 and 2010 show differences in terms of minimum and maximum values. In 1990 the maximum was 0.56, while that of 2010 shows 0.32 (Fig. 11). It means that the forests in 1990 were more vigorous as compared to the forests in 2010. In fact, as it may be noted, the forest in 1990 covered an area of 13,430.50 ha of indigenous forest, which declined to only 2,904.7 ha in 2010, threatening and reducing the potential existence of different species of fauna and flora. The plantations according to the map remained at 8,035ha. Areas of continuous agriculture activities remain non-forests persistently. The water body reduced from 11.50 ha in 1990 to about

6.30 ha in 2010. This reduction in the water body reduced the habitant for water-based fauna and flora.

Results of NDVI analysis for 1990 and 2010 are presented in Figures 11.

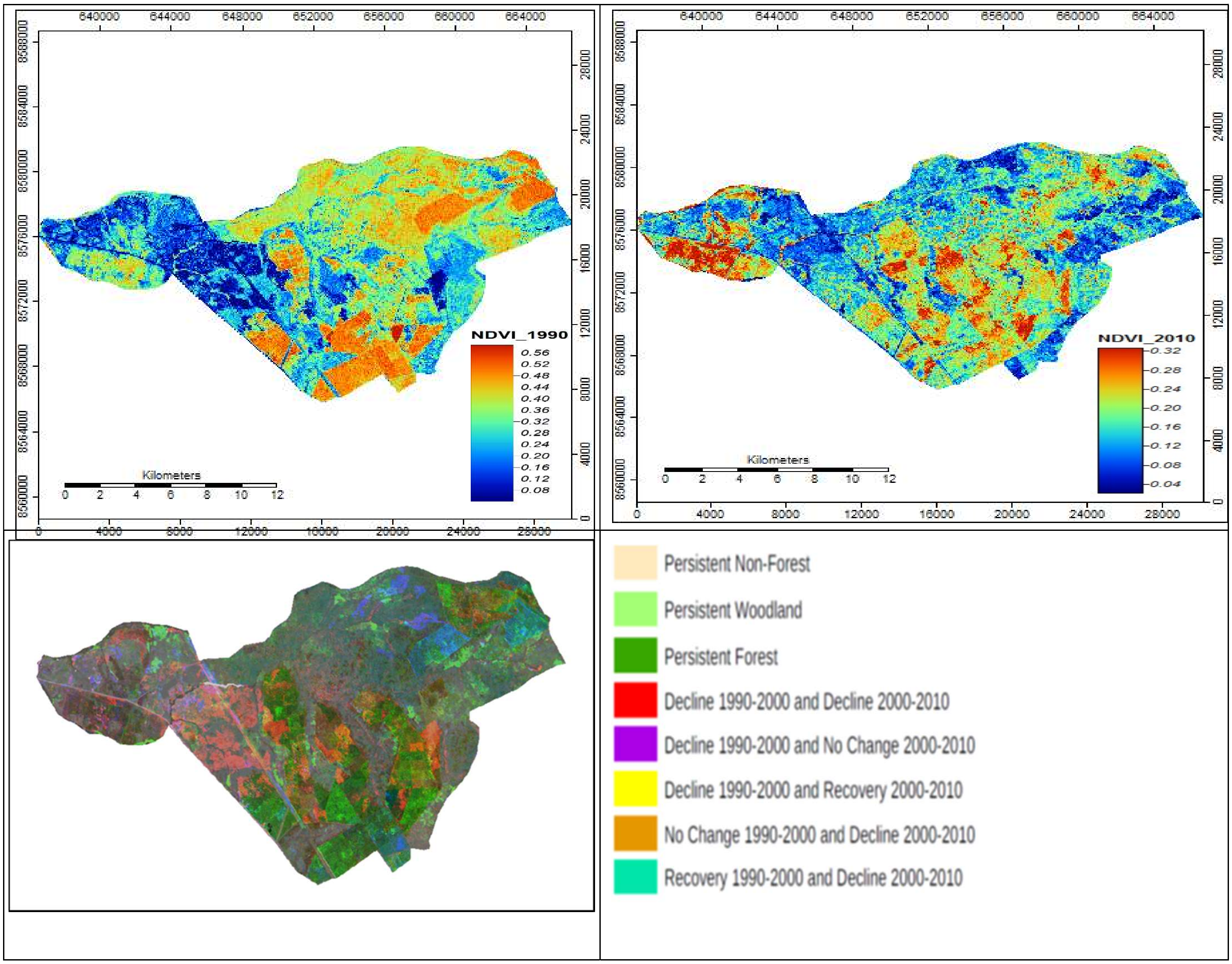

Figure 11: NDVI for 1990 and 2010

Table 11: NDVI Values for 1990 and 2010

Class	**1990**	**Average NDVI**	**2010**	**Average NDVI**
Indigenous Forest	13,430.50ha	0.56	2,902.00ha	0.32
Plantation Forest	8,035.00ha	0.52	8,035.00ha	0.28

The diverse habitats and microhabitats contained in forest ecosystems hold the majority of the world's terrestrial species (Ozanne, et al., 2003). According (CBD, 2001), in the absence of biodiversity there would be no ecosystems and no ecosystem functioning. In addition, there is evidence that complex forest ecosystems are more productive than less diverse ones under the same conditions and that more productive ecosystems are more resilient than less productive ones (Ozanne, et al., 2003). Forests comprising relatively few species supply relatively few goods and services and are highly prone to various catastrophes, including disease and invasion.

It will be noted that in Fig 18, changes over the 20 year period was demonstrated using the NDVI through over laying which showed that the quality of forest and uses of the land varied over the study period.

The loss of biodiversity means the loss not only of the habitat of various species found in the area but also the loss of livelihoods of the people living in and around these forests. As studies have shown (Kalaba & Quinn, 2013) that the forests provided charcoal, fruits, building poles, mushrooms, wild fruits and wild roots which local communities traded in to support their livelihoods in Mwekera.

5 CONCLUSION

The main objective of this study was to investigate the cost of deforestation on ecosystem services. The results from the study demonstrate that Kamfinsa sub-catchment area lost about 572.29ha of forests annually from 1990 to 2010 at the cost of US$243.6 per hectare. As a result of the forest loss (biomass), it was estimated that about 21,938.22 tons of above ground and 6,143.25 tonnes of below carbon per hectare was lost annually. Therefore, a total of 28,081.47 tonnes is emitted from the study area annually. In terms of soil erosion, it was estimated that 1.59 tonnes of soil per hectare were lost annually at an estimated cost of US$57.18 per hectare. Using NDVI as a proxy measure of biodiversity, it was observed that NDVI reduced from 0.56 in 1990 to 0.32 in 2010, an indication of loss of vigour of forests, which is supposed to support both fauna and flora in the study area.

These losses impacted negatively on the supply of ecosystem services including water supply, wood, and non-wood forest products as well as the general livelihoods of the people of Kamfinsa Catchment areas. It is also clear that the monetary estimates underestimates the real costs since there are other benefits derived from the forests.

The study demonstrated that deforestation in Kamfinsa sub-catchment area resulted in costs and therefore there was a need to promote an integrated and holistic approach to addressing the drivers of deforestation in order to ensure continued supply of ecosystem services. In order to reduce these costs, there was need of promoting appropriate agricultural practices and land use planning at local community level. It is recommended that a study based on empirical data should be undertaken in order to improve on the existing data on rates of soil erosion.

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ANNEX 1: ON-SITE SOIL EROSION COSTING CALCULATIONS

Scope and Assumptions

The study made use of data and information that are specific to the Kamfinsa sub-catchment area, specifically Ndola (Figure 7). The methodology for estimating and comparing the on-site costs of soil erosion in crop-covered and non-crop-covered upland areas used the following assumptions.

a. On-site cost of soil erosion is equal to the cost of replacing the lost fertility by restoring the eroded amounts of major soil nutrients- nitrogen (N), phosphorus (P), and potassium (K), plus the labor cost of replacing the topsoil as growing medium, plus the cost of transporting, spreading and incorporating the replacement nutrients and growing medium into the upland soils.

b. The unit values for loss in N, P and K used are those of the miombo woodland derived and reported in a previous study (Figure 7) on similar site by Peter Frost (1996) are appropriate and applicable for this study.

c. Soil erosion rates, quality and composition of topsoil, amount of loss in N, P and K, prices and costs are the same everywhere and are not going to change through time. Erosion rates, quality and composition of topsoil, and losses in N, P, and K are specifically assumed to remain constant regardless of differences in slopes, weather and climate conditions, farmers' practices and the extent and duration of actual erosion in the area.

d. The rate of a Dollar to Kwacha used in the calculations was US$1 = ZWK7

Data Requirements and Sources

Estimation of the on-site costs of soil erosion by the replacement cost method required the following:

a. Quantifying the amount of eroded topsoil which were obtained through this study from the study area by using the RUSLE and GIS methodology;

b. Quantifying the amount of N, P and K contained in 1 ton of undisturbed Miombo topsoil in the study area based on the unit values from 100-g soil samples (Table 4) were adopted and converted to corresponding values for one ton of undisturbed upland topsoil in the area;

c. Quantifying the amount and monetary value of inorganic fertilizers required to replace the lost N, P and K in one ton of eroded topsoil was derived by making use of unit values for soil erosion calculated by using RUSLE and N, P and K content of topsoil (**Table 5**). It also made use of primary data on prices of the inorganic fertilizers, Urea.

Method of estimation

i. Estimating the total weight of N, P and K that were lost through erosion. The procedure used to estimate the total weight of N, P, and K that were lost through soil erosion was adapted from Cho (2010) and Quansah et al (2000). The data used was from previous studies (Table 4).

a. Total weight of Nitrogen (N) in 1 ton of undisturbed miombo woodland topsoil

<u>Assumed Value:</u>

Total N (in %) in 100g of topsoil = 0.068 %
Total N (in g) in 100g of topsoil = 0.068g N

$$\text{Total N (in g)in 1 kg of topsoil} = 0.068\, X \frac{1000 \text{ kg}}{100\text{g}}$$

= 0.68 Kg N in 1 kg of topsoil

$$\text{Total N in 1 kg of undisturbed topsoil} = 0.068\, X \frac{1000 \text{ Kg}}{100\text{g}}\, X\, \frac{1\text{kg}}{100\text{g}}$$

= 0.68 kg N in 1 ton of topsoil

b. Total weight of total Phosphorous (Total P) in 1 ton of undisturbed topsoil

Assumed value

Available P in undisturbed topsoil = 4 ppm

Total P (in %) in 100g of topsoil = 4 ppm

= 4 mg per ton of soil

Available P = 1.28% of Total P (Cho, 2000)

$$\text{Total P (in Kg)in 1 kg of topsoil} = \frac{4 \text{ mg available P}}{0.0128}\, X \frac{1\text{kg}}{1{,}000{,}000\text{mg}}$$

= 0.003125 kg total P in 1 kg of topsoil

Phosphorous is in the form of phosphorous oxide or P_2O_5. Since 1 P_2O_5 has 2P and 5O with a total molecular weight of 142, where 62 is accounted for by 2P, the ratio of P_2O_5 to P is 2.29. Therefore, in P_2O_5 form (in kg) in 1 ton of top soil:

$$\text{Total P (in Kg)in 1 ton of topsoil} = \frac{0.0003125}{1 \text{ Kg topsoil}}\, X \frac{1{,}000\text{kg}}{1 \text{ ton}}$$

= 0.3125 kg total P_2O_5 in 1 ton of top soil

c. Total weight of total Potasium in K_2O form in 1 ton of undisturbed topsoil

Assumed values :

Available K_2O in 100g of topsoil = 17.969 ppm *(Forestry Research, 2015)*

= 17.969 mg per ton of soil

Available K_2O = 10 % of topsoil K_2O

Total K_2O (in kg) in 1 kg of top soil

$$= \frac{17.969 \text{ mg available}}{1.10} X \frac{1\text{kg}}{1{,}000{,}000}$$

= 0.0000163354

Total K_2O (in Kg) in I kg of topsoil:

$$= \frac{0.0000163354 \text{ Kg}}{1 \text{ Kg topsoil}} X \frac{1\text{kg}}{1\text{ton}}$$

= 0.0163 total K_2O in 1 ton topsoil

ii. Estimating the cost per kg of N, P, and K. The respective costs per kg of N, P and K, in 2015 constant prices, in terms of the required volumes of corresponding Urea compound fertilizer. Since fertilizer are not sold as pure N, P and K, the cost of 1 kg of any of the nutrients was made equal to the cost of buying the amount of corresponding fertilizers that would yield a total of 1 kg of the given nutrient.

a. Cost and Volume of Urea required to obtain 1 kg of Nitrogen
Commercial Urea fertilizer has N-P-K ration of 10:20:10

Fertilizer cost for 1 Kg of N

$$\text{Fertilizer Cost for 1 kg of N} = \frac{\text{Price/bag Urea}}{\text{50kg Urea}} X \frac{\text{100kg Urea}}{10}$$

$$= \frac{\text{K250}}{\text{50kg}} X \frac{\text{100kg Urea}}{\text{10kg P}}$$

$$= \text{K50}$$

Volume of Urea that contains 1 kg of N

$$\text{Volume of Urea that contains 1 kg of N} = \frac{\text{K50}}{\text{1 kg N}} X \frac{\text{1 bag Urea}}{\text{K250}}$$

$$= \text{0.2 bag of Urea per 1Kg N}$$

b. Cost and Volume of Urea required to obtain 1 Kg of P

$$\text{Fertilizer Cost for 1 Kg of P} = \frac{\text{k250}}{\text{50 kg}} X \frac{\text{100Kg}}{\text{20kg P}}$$

$$= \text{K25}$$

$$\text{Volume of Urea that contain 1 Kg P} = \frac{\text{K25}}{\text{1 kg P}} X \frac{\text{1 bag Urea}}{\text{K250}}$$

$$= \text{0.1 bag P of Urea per 1kg P}$$

c. Cost and Volume of Urea required to obtain 1 kg of K

$$\text{Fertilizer Cost for 1 Kg of K} = \frac{\text{Price/bag Urea}}{\text{50kg}} X \frac{\text{100kg Urea}}{\text{10kg K}}$$

$$= \frac{250}{\text{50kg}} X \frac{\text{100kg K}}{10}$$

$$= \text{ K50}$$

Volume of Urea that contains 1 kg of K

$$\text{Volume of Urea that contains 1 kg of K } = \frac{\text{K50}}{\text{1 kg N}} X \frac{\text{1 bag Urea}}{\text{K250}}$$

$$= \text{0.2 bag K of Urea per 1 Kg K}$$

iii. Replacement Cost for the lost soil nutrients in one ton of topsoil: The cost of replacing the lost ton of topsoil nutrients was estimated by multiplying the cost of the amount of inorganic fertilizer that contains one (1) Kg of a given nutrient by the amount of nutrient lost in one ton of topsoil due to erosion. The calculations are:

a. Replacement cost for lost N from 1 ton of topsoil:

$$\text{Volume of Urea that contains 1 kg of N } = \frac{\text{K250}}{\text{1 kg N}} X \frac{\text{0.68kg N}}{\text{1 ton topsoil}}$$

$$= \text{K170 per ton of topsoil}$$

b. Replacement cost for lost N from 1 ton of topsoil:

$$\text{Volume of Urea that contains 1 kg of P } = \frac{\text{K250}}{\text{1 kg P}} X \frac{\text{0.3125 P}}{\text{1 ton topsoil}}$$

$$= \text{K78 per ton of topsoil}$$

c. Replacement cost for lost K from 1 ton of topsoil:

$$\text{Volume of Urea that contains 1 kg of P} = \frac{\text{K250}}{\text{1 kg K}} X \frac{\text{0.016 K}}{\text{1 ton topsoil}}$$

$$= \text{K3.75 per ton of topsoil}$$

According to Quansah et al (2000), the NPK contents of the eroded soil can be converted to the forms in which they exist in fertilizers, i.e. N, P_2O_5 and K_2O (kg), respectively, by multiplying the following constants (Quansah, 2000):

Kg N = kg N

Kg P x 2.29 = kg P_20_5

Kg K x 1.2 = kg K_2O

Therefore, one bag of 10:20:10 NPK Compound fertilizer (50kg) cost K250. It is expected that the farmer would replace the lost nutrients in the form 10:20:10 NPK compound fertilizer since 100Kg bag of 10:20:10 NPK contains 10Kg N, 20Kg P and 10Kg K. A 50 Kg bag will contain 5Kg of N, 10Kg of P and 5 Kg of K, giving the amount of nutrients as 20Kg NPK. Since one bag (50Kg) of 10:20:10 NPK fertilizer contains 20Kg NPK, the total amount of NPK in eroded soil can be converted to bags of fertilizer. These results of the soil nutrient loss have been summarized in Table 10.

Table 12: Soil Nutrients in undisturbed Miombo Woodlands

Locality	Parent rock	Depth(cm)	pH (I)	C(%)	N(%)	Exchangeable cations (meq/100 g soil)				BS	TEB	Extr P ppm (2)	Ref
Zambia						Ca	Mg	K	CEC	%	Meq 100g clay		
Kasama	Granite	0-10	4.9a	1.09	-	1.32	0.57	0.10	4.80	41	7	27a	1
		10-20	4.5	0.53	-	0.47	0.29	0.04	3.08	26	4	18	
Kasama	Granite	0-10	4.8a	1.20	0.130	1.64	1.23	0.20	6.42	35	-	6f	2
		10-20	4.5	0.74	0.090	0.57	0.22	0.15	4.89	19	-	2	
Luapula	Precambrian	0-15	4.2c	0.89	0.051	0.16	0.12	0.08	6.92	5	1	<1b	3
	Sediments	40-50	4.3	0.30	0.028	0.24	0.11	0.05	5.16	8	1	<1	
Chingola	Basement	0-10	5.4c	1.90	0.091	0.70	0.55	0.47	9.42	18	4	-	4
	Complex	10-45	5.2	0.52	0.029	0.05	0.11	0.36	5.20	10	1	-	
Ndola	Basement	0-15	5.2b	0.86	0.068	0.15	0.24	0.11	3.10	16	7	4c	5
	Complex	15-30	5.0	0.43	0.035	0.70	0.12	0.06	2.80	9	-	-	
Kapiri-mposhi	Quartz-rich	0-14	5.8a	1.20	0.080	2.90	1.00	0.60	5.20	88	76	44a	6
	Gneiss	14-23	6.1	0.50	0.030	2.20	0.80	0.50	3.80	94	18	32	

Source: (Frost, 1996)

Limitation of Study

The main limitation of the study were:

a. The study did not generate its own unit values for amount of loss in N, P, and K per unit volume of eroded topsoil. It used secondary data (Miombo woodlands) for these values albeit these were derived from data on similar site. Thus, there might be issues about direct applicability and comparability with other slopes, sites and soil qualities;

b. Topsoil fertility does not consist of N, P and K only. Fertile topsoil contains organic matter and other nutrients like Ca, Mg and Fe which cannot be restored into the topsoil by replacing only N, P and K. Ideally, soil fertility replacement cost should include the cost of all other soil nutrients and components;

c. The methodology assumes that unit values and resource qualities are fixed. In reality, these variables change through time. Topsoil quality varies from site to site and from one period to the next even in the same site due to soil organic matter dynamics. The amounts of N, P and K lost through erosion are not the same from year to year, from slope to slope, and from one crop cover to the next.

ANNEX 2: MEAN ANNUAL RAIFALL

Month	Amount
January	292
February	170
March	45
April	3.5
May	0.6
June	0.1
July	0.4
August	2.9
September	31.5
October	130.3
November	305.8
December	102.7
TOTAL	**1084.8**