COUPLING BATHYMETRIC SURVEY AND SEDIMENT GEOCHRONOLOGY TO MODEL IMPACTS OF SOIL CONSERVATION PRACTICES IN LAKE NAIVASHA BASIN, KENYA

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A Thesis submitted in fulfilment for the degree of Doctor of Philosophy in Soil and Water Engineering of Jomo Kenyatta University of Agriculture and Technology.

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DECLARATION

I Caroline W. Maina do declare that this thesis is my original work and to the best of my knowledge has not been submitted to any other university for award of any degree.

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DEDICATION

This is dedicated to my Parents, Brothers and Sisters for the cheering and support you have given me throughout my life. This has always given me a cause to keep going.

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LIST OF ABBREVIATIONS AND ACRONYMS

APS	Acoustic Profiling System
ASCII	American Standard Code for Information Interchange
BSS	Bathymetric Survey System
CA	Cluster Analysis
CF	Contamination Factor
CFCS	Constant Flux Constant Sedimentation model
CHIRPS	Climate Hazard Infrared Precipitation with Station data
CIC	Constant Initial Concentration
CLIGEN	CLImate GENerator
CN	Curve Number
CR	Crescent
CRS	Constant Rate of Supply
CRS-Pw	Constant Rate of Supply-Piecewise
DA	Discriminate Analysis
DAV	Depth - Area - Volume
dBD	dry Bulk Density
DEM	Digital Elevation Model
EF	Enrichment Factor
FA	Factor Analysis
FRNs	Fallout Radionuclides
GPS	Global Positioning System
HP	Hippo Point
HRU	Hydrologic Response Unit
ICP OES	Inductively Coupled Plasma Optical Emission Spectrometer
I_{geo}	geoaccumulation Index
LAR	Linear Accumulation Rate
LaSR	Volumetric Lake Sedimentation Rate

LR	Largest Result
M1, M2, M3	Malewa 1, 2, and 3
MAR	Mass Accumulation Rate
MUSLE	Modified Universal Soil Loss Equation
NSE	Nash Sutcliffe Efficiency
PBIAS	Percent Bias
PC	Principal Component
PCA	Principal Component Analysis
RE	Relative Error
RGS	Regular Gauging Station
RUSLE	Revised Universal Soil Loss Equation
SR	Smallest Result
SSY	Specific Sediment Yield
SV	Sediment Volume
SWAT	Soil Water Assessment Tool
SY	Sediment Yield
TE	Trap Efficiency
TIN	Triangulated Irregular Network
TOPAZ	Topography PArameteriZation
TP	Total Phosphorous
USDA-ARS	United States Department of Agriculture- Agricultural Research Service
USLE	Universal Soil Loss Equation
UTM	Universal Traverse Mercator
WATEM/SEDEM	Water and Tillage Erosion Model/ Sediment Delivery Model
WEPP	Water Erosion Prediction Project
WRA	Water Resources Authority

ABSTRACT

Surface waterbodies are sources of ecosystem functions and support various socio-economic activities. However, these waterbodies are threatened by sedimentation which may result in loss of volume and water quality alteration. Thus, the objective of this study was to assess geochronological sedimentation status of Lake Naivasha in Kenya with a view of modelling impacts of conservation practices. This was achieved by conducting bathymetric survey using multifrequency Acoustic Profiling System (APS), sediment coring, geochronological (using ^{210}Pb and ^{137}Cs radionuclide) and geochemical analysis of sediment. Geochemical analysis of P, Al, As, Cr, Cu, Fe, Mn, Ni, Pb and Zn was conducted using Inductively Coupled Plasma Optical Emission Spectrometry (ICP OES). Sources and contamination levels of these elements were established using multivariate analysis and pollution indices such as enrichment factor. The impacts of adopting conservation practices in Lake Naivasha basin were modelled using Soil Water Assessment Tool (SWAT) and geochronological data. The results showed that between July and October 2016, mean depth, volume and surface area of Lake Naivasha was 4.68 m, 722 x 10^6 m^3 and 154.17 x 10^6 m^2, respectively. Sediment cores, dated up to about 140 years, showed that the mean mass sedimentation rate of the lake is 0.32 g/cm^2/yr. The sediment load into lake Naivasha from 1966 to 1996 and 1996 to 2016 was found to be 2.78 x 10^5 and 4.61 x 10^5 tons/yr, respectively. The difference in sediment load indicates increased anthropogenic activities upstream. Elements (P, As, Fe, Mn and Zn) in Lake Naivasha sediment were found to be from both natural and anthropogenic sources while the rest (Al, Cr, Cu, Ni, and Pb) were from natural sources only. Adoption of filter strips and terraces, as conservation practices on critical and agricultural land, would reduce cumulative sediment load into Lake Naivasha by up to about 30% and 27%, respectively. The impact of these practices increased by up to 11% when implemented in all sub basins with agriculture. It was found that with use of multifrequency APS in combination with dated sediment cores, it is possible to assess the sedimentation status of a natural lake that has no comparable bathymetric data or known pre-impoundment layer like man made reservoirs. Coupling bathymetric survey, sediment geochronology and SWAT model can inform the choice of plausible intervention measures that would reduce sedimentation of waterbodies.

CHAPTER ONE

INTRODUCTION

1.1 Background Information

Water demand has been on the rise globally as population increases. As a result, surface waterbodies such as rivers, lakes and reservoirs are highly exploited as fresh water sources for domestic, industrial and agricultural purposes. However, the functions of these waterbodies are threatened by sedimentation (Tigrek and Aras, 2012; Mulu and Dwarakish, 2015). According to Odhiambo and Ricker (2012) soil erosion and associated sedimentation, threaten sustainable use of surface water resources through loss of volume storage capacity and conveyance of pollutants to receiving water bodies. Soil erosion from the catchment and at the shores of rivers, reservoirs and lakes leads to siltation of waterbodies (Lachhab *et al.*, 2015). The siltation results in increased turbidity, clogging of hydraulic structures and reduced factor of safety for flood water retention. Further, operation and maintenance of engineering facilities and environmental functions are also affected by sedimentation of the waterbodies (Zarris and Lykoudi, 2017). Studies conducted by Schleiss *et al.* (2008) and Rabee *et al.* (2011) indicate that the annual storage capacity loss is higher than the increase of storage capacity achieved by constructing new reservoirs worldwide. Other studies by Basson (2010), Rakhmatullaev *et al.* (2011), Schleiss *et al.* (2016) and Zarris and Lykoudi (2017) indicate that the annual loss in reservoir storage capacity due to sedimentation is about 0.5-1%.

Research work on storage capacity loss are further supported by findings from studies carried out in different African countries. For instance, a study conducted in North Africa reservoirs by Lahlou (1996), gave an estimated annual reservoir storage loss as 0.5%, 0.7% and 1.2% for Morocco, Algeria and Tunisia, respectively. In addition, some reservoirs in Sudan have been reported to have an annual storage capacity loss of 1.25%

(Shahin, 1993). A report by Hunink *et al.* (2013) on reservoir sedimentation in the Upper Tana Basin in Kenya, indicates that reservoir capacity loss of about 10 – 30% could be realised by 2010. Further, studies in the Upper Tana Basin indicate that in Kenya, Masinga dam, had lost 10.1% of its capacity over 29 years from 1980 while Kamburu lost 14.7% of its capacity in 27 years between 1983 - 2010 (Hunink and Droogers, 2011). Almost similar results on Masinga dam were reported by Saenyi (2002) on loss of reservoir storage capacity where a 10.1% and 13.7% reduction in storage capacity in 1981 and 2000 were estimated using WEPP model. The Masinga reservoir sedimentation was attributed to accelerated soil erosion in Masinga catchment.

According to Bunyasi *et al.* (2013), soil erosion is a severe problem affecting agricultural lands within high rainfall areas as well as in semi-arid areas. Obando (2005) indicated that high rainfall areas cover about 17% of Kenya's total land area. These high rainfall areas support about 80% of the country's rural population. On the other hand, 60% of livestock and wildlife are found in semi-arid land. Depending on the soil types in these areas, high soil erosion is experienced particularly on steep slopes thereby increasing sediment fluxes into water bodies. This leads to an increase in reservoir sedimentation (Wambua and Kithiia, 2014). Soil erosion leads to loss of reservoir storage capacity and additionally causes water quality degradation (Fan and Morris, 1992; Sohoulande, 2018).

Water quality within the reservoir is affected by the enrichment of nutrients and agrochemicals carried by the sediments (Yuan *et al.*, 2011; Bhuiyan *et al.*, 2015; Skordas *et al.*, 2015). Agricultural activities have been reported as a major source of water pollution in Kenya (Ouma *et al.*, 2013). Many forested catchments in Kenya have been converted to farmlands even on the steep slopes and this has resulted in more runoff (Wilschut, 2010). Increased runoff leads to more soil being eroded within the catchment and eventually deposited into the receiving waterbodies. Further, land use changes and activities have great effects on lake/reservoir water quality (Kithiia, 2012).

One such a lake that is characterized by extensive agricultural activities within its basin is Lake Naivasha, Kenya.

Lake Naivasha basin is characterized by agricultural land uses where large areas have been converted into farmlands even on steep slopes (Becht and Harper, 2002; Kitaka *et al.*, 2002; Tarras-Wahlberg *et al.*, 2002; Harper *et al.*, 2011). The high agricultural activities within Lake Naivasha basin have led to increased soil loss (Tarras-Wahlberg *et al.*, 2002) which may negatively impact on the aquatic environment. Such environments have been linked to diverse types of species and environmental processes, flow of nutrients and sedimentation (Ouma *et al.*, 2013). Yatich *et al.* (2009) reported that the increased land use change within a catchment can be linked to sediment loading into lakes and reservoirs. The sediments may be highly contaminated with nutrients such as phosphorous and other agrochemicals. For instance, the increased nutrient availability in Lake Naivasha has been attributed to intensive farming in the highlands of the catchment by Stoof-Leichsenring *et al.* (2011). The same study reported that increased population and flower farming on the southern shore of the lake resulted to its eutrophication. In cases where the lake is used to supply drinking water, the quality of the water may be lowered leading to increased cost of water treatment. Thus, there was need to investigate sedimentation status of Lake Naivasha by conducting bathymetric survey and establishing sources of sediment within the basin with a view of modelling impacts of conservation practices on lake sedimentation.

1.2 Statement of the Problem

Lake Naivasha basin has experienced an increase in anthropogenic activities which could have an impact on the upstream and downstream areas. In the upstream, farmers lose their top fertile soils to agents of erosion. In the downstream, Lake Naivasha bathymetric characteristics are affected by sedimentation and the water quality is impaired by the chemical loading. The rate of sediment and chemical load is not well known. Almost all gauging stations existing within Lake Naivasha basin do not monitor the sediments load into the lake. Thus, spatial-temporal sources and sinks of sediment within the basin and the lake are unknown. Also, the current bathymetric conditions of Lake Naivasha are unknown and its management is based on bathymetric findings for survey conducted back in 1983 and reported by Ase *et al.* (1986). It is observed that there are no comparable bathymetry surveys of Lake Naivasha which could provide information about sediment distribution and long term sedimentation rate of the lake. Rupasingha (2002) reported a sedimentation rate of 0.5 cm/yr between 1957 and 1990. However, he did not give spatial-temporal distribution of this rate. Similarly, most studies on Lake Naivasha sedimentation have focused on short-term sediment measurements which considers suspended sediments only. This lack of long-term sedimentation data limits application of models in decision making such as establishing critical areas that could be prioritized for conservation and their impacts on lake sedimentation.

1.3 Objectives

1.3.1 Main Objective

The main objective of this study was to combine bathymetric survey and sediment geochronology to establish sedimentation status and to model impacts of soil conservation practices in Lake Naivasha basin.

1.3.2 Specific Objectives

The specific objectives of this study were to:

i. Evaluate geochronological sedimentation status of Lake Naivasha through bathymetric survey and sediment coring,
ii. Determine geochronological source and contamination level of heavy metal and nutrients in lake sediments,

iii. Determine the impacts of adopting soil conservation practices on critical areas and agricultural land in reducing sedimentation of Lake Naivasha.

1.4 Research Questions

i. What is the sedimentation status of Lake Naivasha?
ii. What are the geochronological sources and contamination levels of heavy metals in Lake Naivasha sediments?
iii. How will selected conservation practices in critical areas and agricultural land impact sediment loading into Lake Naivasha?

1.5 Justification

A study on quantity and quality of sediment deposited in a reservoir aids in undertaking effective reservoir and basin management (Majumdar, 2015). Lake Naivasha basin lies within intensive agricultural catchments (Harper *et al.*, 2011). As a result, the lake is threatened by sedimentation (Tarras-Wahlberg *et al.*, 2002). Historical data on discharge and in some cases short-term suspended sediment data has been used in simulating basins processes, but there is a continued increase in reservoir sedimentation as a result of the increased anthropogenic activities within the basin (Hunink *et al.*, 2013). Hence, reservoir-based studies for Lake Naivasha would be useful in establishing or reconstructing long-term sediment yields. Long-term sediment yields records are important in estimating changes in the lake and its catchment over the years and thus giving rich history of human-nature interaction (Yanhong *et al.*, 2004; Ntakirutimana *et al.*, 2013; Al-Mur *et al.*, 2017; Sahli *et al.*, 2017). In addition, investigation on sediment quality is also an environmental concern since sediment act as a sink for water quality contaminants and a source of contaminants to water in the lake/reservoir and biota (Bing *et al.*, 2013; Elkady *et al.*, 2015). Within the study basin, little has been done to understand the extent to which land use change impacts on sediment fluxes in the environment. Further, a relationship between sedimentation rates of Lake Naivasha and different land use practices over time has not been established. This relationship is very crucial in managing sediment yield delivery into the reservoir.

According to Zanjani-Jamshidi and Saeedi (2017) historical information on sediment deposition can be derived from analysis of lake bottom sediments. A study by Bing *et al*. (2013) and Purushothaman *et al*. (2012) reported that lake sediments are useful in assessing the health status of aquatic ecosystems. Also, assessment of lake sediments is used during environmental surveys to aid in establishing changes in sediment characteristics input with time (Dai *et al*., 2007). According to studies conducted by Al-Mur *et al*. (2017), Sahli *et al*. (2017) and Zanjani-Jamshidi and Saeedi (2017) the authors reported that investigation along the sediment core provides a reliable historical natural and anthropogenic impacts on nutrients and heavy metals accumulation in sediments over time.

Dating of sediment core using the natural or artificial Fallout Radionuclides (FRNs) is normally used to assign time on the sediment. The most common FRNs used in assessing recent sedimentation of reservoirs through dating of the sediment core include Cesium-137 (^{137}Cs), Lead-210 (^{210}Pb) and Beryllium-7 (^{7}Be) nuclides (Yanhong *et al.*, 2004; Putyrskaya *et al.*, 2015). From the dated sediment core, sedimentation rate of the reservoirs can be understood, and this can be linked to noted changes of anthropogenic activities within the catchment. The information generated from this kind of a study aids in generating useful information to catchment managers and decision makers in land use and reservoir management. The data is also useful in calibrating models which would aid in identification of critical areas within the basin that should be prioritized for conservation measures. The extent of soil loss that results to lake/reservoir sedimentation varies spatially within basins. As a result, it is important to establish soil conservation strategies that will focus on the spatial disparity of soil loss within the basin.

1.6 Scope and Limitations of the Study

While this study sought to determine reservoir sedimentation using bathymetric survey, sediment coring, chronological studies and modelling, suspended sediments within the water bodies was not measured. In addition, the historical discharge data of the inflow streams was used in the model since direct measurement was not undertaken. Due to the cost involved and availability of equipment, only two sediment cores obtained from Lake Naivasha were analysed using ^{137}Cs and ^{210}Pb radionuclides for chronological investigation. Also, the sampling points within the study area were sparsely distributed to have manageable sediment samples. In modelling soil conservation practices an assumption was made that there are no existing filter strips and terraces in the basin.

CHAPTER TWO

LITERATURE REVIEW

2.1 Soil Erosion and Sediment Transport

Although soil erosion is a natural process, anthropogenic activities speed up the process. Worldwide, human induced soil erosion resulting from agricultural activities, clearing of forests, mining and construction accounts for about 75 billion tons of soil per year. This rate is about 13 – 40 times as fast as the natural rate of erosion (Surjit *et al.*, 2015). Dutta (2016), reported that about 80 % of agricultural land suffers from moderate to severe erosion globally. Studies show that soil erosion has been increasing in the recent past and thus poses a serious threat towards food production (Pimentel and Burgess, 2013). Much of this increase in soil erosion can be attributed to anthropogenic factors such as deforestation; overgrazing, mining, construction and poor tillage methods (Greig *et al.*, 2005; Gellis *et al.*, 2006; Perovic' *et al.*, 2013). Soil erosion is also closely related to nutrient depletion within agricultural land. The loss of nutrients negatively affects agriculture productivity and hence contributing to food insecurity (Bosco *et al.*, 2015).

Accelerated soil erosion is a fundamental problem that greatly impacts on economics and environmental issues worldwide (Alemaw *et al.*, 2013). Cabahug and Villanueva (2014) pointed out that sediment transport depends on particle size, discharge, shear stress, flow velocity and energy gradient. On the other hand, Hudson and Mossa (1997) noted that when the source of sediment is fine grained soil or alluvial clay, the sediments remain mostly in suspension. Munthali *et al.* (2011) reported that sediment transportation is an indicator of erosion processes in the watersheds and it leads to loss of top fertile soils. As a result, soil erosion leads to loss of soil nutrients which leads to reduction of soil productivity and subsequently a decline in crop yields.

As soil loss and sediment transportation take place, agricultural chemicals, pathogens and nutrients such as phosphorous get into water bodies consequently lowering reservoir water quality (Dupas *et al.*, 2015). If eroded and transported soils have nutrients and contaminants, surface water pollution and eutrophication is experienced. As a result, aquatic life and human health is threatened (Ouma *et al.*, 2013). Transportation may be caused by splash erosion in suspension or by saltation and rolling (Kinnell, 2010).

Walling (2009) reported that, a strong correlation between rates of erosion, rainfall intensity, river drainage and sediment transport exist. The main physical processes within sediment cycle, i.e. erosion, transport and deposition were also reported to be nonlinear and that they are highly affected by hill slopes as well as catchment waterway management.

2.2 Sediment sources and sinks

Sediment is a naturally occurring material that is broken down by processes of weathering and erosion. The material is subsequently transported by water, wind, or by the force of gravity acting on the particles (Morris and Fan, 1998). There are two broad categories of sediment sources namely upland sediment sources and channel sediment sources. The sediment sources within a on the upland areas include land use within the basins, agricultural areas, mines, construction sites and roads. On the other hand, channel sediment sources include gullies and ditches, streambeds and banks where streambanks erode during high flows . In the upland areas, erosion occurs due to sheet erosion, rill, gullies and mass movements. On the other hand, construction of dams and impoundments acts as sediment sink within a basin which reduces the supply of sediment to downstream reaches of a river. Also, channels, gullies, depressions and forests are sinks of sediment (Owens *et al.*, 2010). Ketelsen *et al.* (2013) reported that vegetation, check dams, small irrigation dams amongst others act as buffers that reduce transportation, sediment yield and reservoir sedimentation. The total amount of erosion material transported from a basin is considered as sediment yield (Munthali *et al.*, 2011).

Sediment yield acts as an indicator of erosion susceptibility within the catchment and aids in determining the amount of sediment that may find its way to the reservoirs or any other point of interest in a watershed (Verstraeten and Poesen, 2001a). Sediment yield is influenced by the extent of catchment sediment trapping through hill slopes, check dams, gullies, reservoir and micro topographic features. The long-term sediment yields have been used in sizing of storage reservoir and estimation of reservoir life span (Foster and Walling, 1994; Lim *et al.*, 2005; Majumdar, 2015).

In addition, sediment yield has been reported by Verstraeten and Poesen (2001a) to be a useful indicator of reservoir sedimentation since it refers to the amount of sediment getting out of the catchment. According to Morris and Fan (1998) and Verstraeten and Poesen (2001b) reservoir survey data represents more reliable measure for long-term basin sediment yield over large areas. The reservoir/lake deposited sediment is an indicator of suspended and bedload sediment transport from a catchment. According to Kokpinar *et al.* (2015) traditional sediment sampling techniques in rivers usually underestimate the total sediment yield since these techniques are mostly limited to analysis of suspended sediment load. Hence, use of traditional sediment sampling techniques could lead to under estimation of sediment deposition in lakes/reservoirs. As a result, there is need of assessing lake/reservoir sedimentation using techniques such as multifrequency Acoustic Profiling System which combine both the suspended and bedload sediment.

2.3 Lake Sedimentation

Lake sediments are comprised mainly of clastic material which include sediment of clay, silt, and sand sizes. Also, organic debris, chemical precipitates, or combinations of these materials are in the sediment (Verschuren *et al.*, 1999). Lake sediment is characterized by two basic components which include allochthonous and autochthonous material. The allochthonous materials originates from lake basin and is transported to the lake by rivers and streams, overland flow and/or eolian activity (Fukushima *et al.*, 1987). On the other hand, autochthonous material, is produced within the lake itself. The autochthonous sediment is either due to inorganic precipitation within the water column or it is biogenic (Verschuren *et al.*, 2002). According to Mwamburi (2019) when autochthonous matter is dominant, lake sediments are either carbonate-rich or silica-rich. This is usually due to the accumulation of siliceous diatoms. For autochthonous sediment, organic carbon content may reach 20–25% (Verschuren *et al.*, 1999). According to Fukushima *et al.* (1987) condition of water movement and depth of the lake affect autochthonous deposited sediment. In addition, the type of phytoplankton species affects the autochthonous sedimentation (Takamura *et al.*, 1984).

On the other hand, according to Sohoulande (2018), allochthonous sedimentation is usually affected by soil erosion within the catchment, rate of transportation and rate or mode of deposition. At the head water area, coarse sediments are deposited while density currents transfer finer sediments further into the lake (Szmytkiewicz and Zalewska, 2014). The functions of lakes and reservoirs are threatened by the rapid loss of storage capacity due to sedimentation. This results to loss of intended services despite the cost incurred during development. Hence, there is need to tackle on- and of-site threats of erosion through catchment-based management strategies.

Sediment inflow into a water body, negatively impacts on the water quantity and water quality (Pilgrim *et al.*, 2015). The water quantity is affected by reducing the storage volume of the lake /reservoir or stream channel. The sediment materials may be loaded with nutrients such as nitrogen and phosphorous, agro-chemicals and heavy metals which affects the water quality (Busari *et al.*, 2014). Understanding reservoir sedimentation is of fundamental significance in hydro systems engineering.

2.3.1 Sediment Distribution in Reservoirs

The study of sediment distribution within the reservoir is useful to designers and planners of most hydraulic structures (Issa *et al.*, 2013a). However, most studies on reservoir sedimentation have focused on predicting and reducing sediment infilling rate using models but little has been done to determine the sediment distribution and pattern within the reservoir or lake. The pattern of sediment distribution is controlled by either primary or secondary processes. The primary process refers to initial sediment deposition while secondary process refers to sedimentation that takes place after resuspension and redistribution of the previously deposited sediment (Shotbolt *et al*., 2005).

In some cases, deltas act as depositional or erosional zones to lakes or reservoirs and are sources of sediment reworking in lakes or reservoirs. According to Shotbolt *et al.* (2005) reservoirs are subject to alternating periods of deposition and erosion due to changes in stream inputs and reservoirs water levels. Sediment deposition pattern depends on factors such as; characteristics of the sediments plus quantity of moving sediment and stream flow (Morris and Fan, 1998; Issa *et al.*, 2013a). The coarse particles usually settle first thus forming the delta while the fine sediment is transported by turbid density currents or non-stratified flow resulting to bottom set beds (Issa *et al*., 2013a). Further, hydrodynamics play an important role in sediment redistribution and contaminant movement in shallow lakes (Ndungu *et al.*, 2015). As a result, there was need in this study to assess the lake/reservoir bed profile as well as sediment distribution. This can be achieved using various techniques such as the use of spud bar or bathymetry survey. According to Shotbolt *et al.* (2005) assessment of sediment distribution within a reservoir is also paramount prior to the collection and selection of suitable cores for environmental analysis.

2.3.2 Determining Reservoir Sedimentation

In assessing sedimentation status of a lake or reservoir, past and current bathymetric surveys can be compared and the actual sedimentation rate be determined (Zarris and Lykoudi, 2017). According to Rakhmatullaev *et al.* (2011), volume and surface area differences derived from results of multiple surveys for individual reservoirs provide estimates of the capacity loss over time due to sedimentation. Thus, to determine sedimentation within a lake or reservoir, bathymetric surveys conducted after a span of years provides significant insights such as estimating the overall sediment distribution, rate and thickness within the reservoir.

According to Jakubauskas and DeNoyelles (2008), sediment thickness and volume can be estimated using topographic maps, sediment coring, spud bar and multifrequency acoustic systems. Some approaches such as the use of sediment coring and spud bar method provide a limited representation of actual sedimentation rates, distributions and patterns since they are limited to a few point samples (Dunbar *et al.*, 1999; Dunbar *et al.*, 2003; Odhiambo and Boss, 2004; Bennett *et al.*, 2013). Hence, acoustics have been used in bathymetric surveys to establish water depth and in some cases both water depth and sediment thickness.

To establish information on sediment thickness using acoustics the bathymetric survey findings are compared with the topography at the time of dam construction or with the previous bathymetric surveys conducted for the reservoir or lake. However, according to Dunbar *et al.* (1999) for proper comparison of bathymetric survey results, a 10 years span between the surveys is paramount. Bathymetric surveys are also used in quantifying the magnitude of bottom sediment erosion, deposition and redistribution processes resulting from storm events. This is usually possible by comparing results of bathymetric surveys conducted before and after the storm events (McAlister *et al.*, 2013; Lachhab *et al.*, 2015). However, for proper comparison of different bathymetric survey results, there is need to use the same survey lines, equipment and procedures, otherwise the computed volume loss may not be comparable (Solis *et al.*, 2012; McAlister *et al.*, 2013).

According to Dunbar *et al.* (1999), overstated pre-impoundment reservoir capacity leads to inaccurate estimation of reservoir sedimentation rates. Odhiambo and Boss (2004) also reported that estimated reservoir sedimentation rates are affected by over-reliance on Universal Soil Loss Equation. Thus, to accurately predict reservoirs and/or lakes sedimentation rate, there is need for a methodology that can aid in establishment of sediment thickness throughout the lake or reservoir. One such method involves the use of multifrequency Acoustic Profiling System (APS) which allows simultaneous determination of water depth and sediment thickness (Dunbar *et al.*, 1999; McAlister *et al.*, 2013).

2.3.3 Multifrequency Acoustic Profiling System in Sedimentation Survey

The system operates on different frequencies with higher frequency being useful in determination of water depth while the low frequency has more energy that penetrates the recently deposited sediment. For instance, one multifrequency APS has 200 kHz, 50 kHz and 12 kHz frequencies. In this case, the 200 kHz enables determination of sediment surface thus giving the water depth. The fine-grained sediment is mapped using 50 kHz signal while the 12 kHz frequency can map up to about 50 m of coarse-grained Sediment (Dunbar *et al.*, 1999; Moriasi *et al.*, 2018). Thus, with use of 50 and 12 kHz frequencies, the thickness of deposited sediment can be determined. The advantage of this method over the traditional bathymetric survey approaches is that it does not depend on previous surveys to determine storage capacity loss or sediment thickness (Dunbar *et al.*, 1999). To improve on the accuracy of the multifrequency APS bathymetric survey results, the method can be validated by collecting sediment cores using vibe coring technique or by using spud bar approach (Odhiambo and Boss, 2004; McAlister *et al.*, 2013; Yutsis *et al.*, 2014).

According to McAlister *et al.* (2013) the use of multifrequency survey technique aids in assessing the history of sediment delivery into the reservoirs and in establishing the effectiveness of adopted conservation practices. Bathymetric surveys conducted years after adoption of catchment conservation practices can be used to quantify the impacts of management practices (Odhiambo and Boss, 2004). From the established information, sedimentation dynamics of a reservoir and/or lake can be established. The sediment cores collected are studied for their core stratigraphy characteristics and are used in verifying temporal consistency of environmental records. This is achieved through chronological analysis of sediment cores where dates are assigned to various layers. Bathymetric surveys play a key role by chronological analysis in estimating sedimentation rates within catchments. The method is robust and has few field surveys compared to other methods such as sediment rating curves which require continuous sediment gauging (Langland, 2009).

When bathymetric surveys are calibrated using sediment cores to measure sediment thickness, an accurate reservoir capacity and long-term water volume loss can be determined in a single field survey (Dunbar *et al.*, 1999). Combination of vibe coring, multifrequency APS and geochronological interpretation using ^{137}Cs radionuclide has been exploited previously by Dunbar *et al.* (1999), Dunbar *et al.* (2003) and Dunbar *et al.* (2013) to assess sediment impoundment volume in reservoirs. However, in natural lakes such as Lake Naivasha which has no known pre-impoundment status, little has been studied on the use of combined methodology in establishing their sediment volume and sedimentation status over a given period.

2.4 Chronological Sediment Survey

Chronological studies are useful in determining whether buried substances are migrating or degenerating. The radionuclide concentration can be determined with a single site visit thus reducing the time required in the field for data collection. In addition, chronological sediment surveys are also cost effective than direct soil loss measurements (Huh and Su, 2004). Use of radionuclides such as Cesium - 137 (^{137}Cs), Lead – 210 (^{210}Pb) and Berylium 7 (^{7}Be) helps in estimating soil loss and spatial sediment redistribution within a catchment (Walling *et al.*, 2014). Soil loss and sediment redistribution study is important in assessing the onsite soil degradation and offsite sediment problem (Porto and Walling, 2012). This approach provides information on medium and long-term average rates of soil redistribution with minimal field visits.

Despite different successes in use of ^{7}Be, ^{137}Cs and ^{210}Pb ex, the methods have not been widely investigated in developing countries especially in Kenya.

2.4.1 Berylium 7

The Berylium 7 (^{7}Be) radionuclide is produced in the stratosphere and troposphere when oxygen and nitrogen spallate (Guzmán *et al.*, 2013). According to Marestoni *et al.* (2009) 95% of total ^{7}Be fallout is through precipitation. The nuclide is mostly detectable near the surface of the sediment samples and has a half-life of 53.3 days. As a result, the nuclide activity is counted soon enough after a sample collection (Zapata and Agudo, 2000). Further, due to the short half-life of the nuclide, it is not found below 20 mm of any soil and sediment, hence it is useful for surface sediments recovery (Wilkinson *et al.*, 2006).

The ^{7}Be radionuclide was reported by Marestoni *et al.* (2009) to be useful in identifying the newly deposited sediment. It is also used when assessing erosion over a short period. The ^{7}Be total inventory and its vertical profile distribution is also used in assessing sedimentation in an area. An increase in depth of sediment is characterized by an exponential decline in ^{7}Be activity concentration. In cases where ^{7}Be is used to study soil erosion, a loss of ^{7}Be from surface soil indicates that soil erosion has taken place (Mabit *et al.*, 2014b). The ^{7}Be nuclide is also useful in documenting soil erosion and sedimentation rates associated with rainfall events. Further, the ^{7}Be nuclide are useful in assessing the effectiveness of recent soil conservation practices. Two main assumptions govern the application of ^{7}Be in soil erosion and sedimentation studies. These assumptions consider that the input of ^{7}Be activity is spatially uniform, and, that any pre-existing ^{7}Be is uniformly distributed across the area under investigation (Marestoni *et al.*, 2009). According to Putyrskaya *et al.* (2015) the nuclide is useful as a time marker since it assigns the initial time at the top of sediment core. This aids in dating the successive layers using ^{210}Pb and ^{137}Cs nuclides.

2.4.2 Cesium - 137

Cesium - 137 (^{137}Cs) is an artificial radionuclide which originated from nuclear weapons testing (Ritchie and McHenry, 1990; Walling, 1998). The nuclide was first detected in early 1950s and was at peak early 1960s (Stefano *et al.*, 1999; Mabit *et al.*, 2014c) before the nuclear test ban treaty. The layer with peak activities of ^{137}Cs is assigned to year 1963 when the world experienced maximum fallout. In the mid-1970s, the ^{137}Cs declined to very low levels. According to Poreba (2006), ^{137}Cs has a half-life of about 30 years. Gellis *et al.* (2006); Mahawatte and Abeynayake (2010) and Walling (1998) reported that clay and organic matter in soils have a strong adsorption of ^{137}Cs. The nuclide is strongly adsorbed to fine particles and as a result it is useful in studying the sediment chronology (Patrocinio and Andrello, 2009; Dercon *et al.*, 2012). The ^{137}Cs concentration, along the sediment profiles give the deposition history that can be interpreted to assign calendar dates (Putyrskaya and Klemt, 2007)

The use of ^{137}Cs in sedimentation studies provides a great potential in data scarce regions (Porto *et al.*, 2014). Although, possibilities of using ^{137}Cs in assessing soil erosion and sedimentation in developing countries was discussed in 1995 (IAEA, 1998) very little work using the ^{137}Cs has been done in most African countries. In the southern hemisphere, the ^{137}Cs concentration is low since nuclear weapon testing was mainly on the northern hemisphere (Ritchie and McHenry, 1990; Walling, 1998; Stefano *et al.*, 1999; Dercon *et al.*, 2012). As a result, more time is required to detect ^{137}Cs durin analysis of soils and sediments. Hence, in the southern hemisphere, ^{137}Cs application is limited in soil erosion and deposition determination, since the quantities measured are very low or in some cases below detection limit. In such areas erosion and sedimentation rates can be assessed by combining ^{137}Cs and natural radionuclide such as ^{210}Pb.

2.4.3 Lead – 210

Lead – 210 (^{210}Pb) is a natural radioactive form of lead found in small quantities in most soils as part of uranium decay series and has a half-life of 22.3 years (Lubis, 2006; Porto *et al.*, 2006; Zhang *et al.*, 2017b). According to Guzmán et al. (2013) and Putyrskaya et al. (2015) the ^{210}Pb is usually derived from radioactive decay of Radon 222 (^{222}Rn) gas daughter of Radium 226 (^{226}Ra). The ^{226}Ra occurs naturally in soils and rocks and generates ^{210}Pb that is in equilibrium with its parent material (Shakhashiro and Mabit, 2011). When in the soil, there is an upward diffusion of ^{222}Rn in small quantities. Hence ^{210}Pb is introduced into the atmosphere and its fallout leads to an increase of the ^{210}Pb radionuclide to the soil surface which eventually interferes with the equilibrium of the parent ^{226}Ra (Mabit *et al.*, 2014a). This fallout component is referred to as unsupported or excess ^{210}Pb since it cannot be accounted for by the in-situ parent.

The concentration of ^{210}Pb is greatest at the surface because of its continuous replenishment at the soil surface and has an exponential reduction in activity below the surface (Zhang *et al.*, 2017b). Decreasing trend in fallout ^{210}Pb with time is caused by radioactive decay where deeper levels in a core correspond to earlier times and this manifest itself with decreasing concentration with depth concentration (Jeter, 1999). According to Mabit *et al.* (2014a) excess ^{210}Pb (^{210}Pbex) is assumed to be constant from year to year and as a result, the downcore variation of ^{210}Pbex activity signifies changes in sedimentation rate. The ^{210}Pb activity declines slowly with depth when high sediment accumulation rates are experienced and vice versa (Zhang *et al.*, 2017b).

According to Jiniun *et al.* (2006), Benmansour *et al.* (2014) and Mabit *et al.* (2014a) ^{210}Pb aids in estimating soil redistribution over 100 years. Further, characterization of changes in sediment input into a lake or reservoir can be achieved using ^{210}Pb fallout radionuclide in sediment dating along the core (Dai *et al.*, 2007; Yang and Turner, 2013; Yusoff *et al.*, 2015; Zalewska *et al.*, 2015). According to Szmytkiewicz and Zalewska (2014) the excess/unsupported ^{210}Pb is useful in dating of the various layers in sediment and establishing sediment accumulation rate either as Mass Accumulation Rate (MAR) or Linear Accumulation Rate (LAR).

This is achieved by employing different models such as; Constant Flux Constant Sedimentation (CFCS) model, Constant Initial Concentration (CIC) model and Constant Rate of Supply (CRS) model. The CIC model assumes that initial ^{210}Pbex concentration from the atmosphere to the deposited sediment is constant through time (Appleby and Oldfield, 1978; Putyrskaya *et al.*, 2015). From CIC model, the age is calculated using Equation 2.1.

$$t_x = \frac{1}{\lambda} \ln \frac{C_0}{C_x} \tag{2.1}$$

where,

t_x = Age of sediment layer (years)

λ = ^{210}Pb Radioactive decay constant (yr $^{-1}$)

C_0 = Initial $^{210}Pb_{ex}$ activity concentration (Bq/g)

C_x = Present $^{210}Pb_{ex}$ activity concentration (Bq/g)

x = Depth (cm)

The CIC model is easily applied since an accurate knowledge of the total unsupported ^{210}Pb inventory is not required for the date to be determined. The CIC model was further improved into CFCS model which assumes that the ^{210}Pbex flux on sediment surface and the sedimentation rate are constant In CFCS model it is assumed that a constant flux of ^{210}Pb is experienced and that the rate of sediment deposition is also constant. Hence, a vertical distribution of ^{210}Pbex in sediment is computed using Equation 2.2 after Putyrskaya *et al.* (2015).

$$C_x = C_0 \, . e^{-\lambda \, w / R_S} \tag{2.2}$$

where,

w_x = Cumulative weight of sediments above depth x (g/cm^2)

R_S = Mass sedimentation rate (g/cm^2/yr)

A plot of ^{210}Pb activities against the cumulative weight accounts for sediment compaction with depth. When CFCS model is used, sedimentation rate (R_S) is determined from the slope of the line that is derived from linear regression of ln^{210}Pb and the depth layer. After the sedimentation rate is established, the age of sediment layers is calculated using Equation 2.3 given by Robbins *et al.* (1978).

$$t_x = \frac{w_x}{R_S} \tag{2.3}$$

In catchments which have been influenced by anthropogenic activities and environmental changes, sediment yield is also affected. As a result, the initial ^{210}Pb activities in the sediment and sedimentation rates may not be constant hence CIC And CFCS are limited in sediment dating. As a result, CRS model which accounts for variations in sedimentation rate should be applied. The model assumes deposition of the same amount of excess ^{210}Pb on sediment per given time and a variable sedimentation rate. The CRS model automatically accounts for the sediment compaction with depth. When calculating sediment age corresponding to a certain depth (Equation 2.4) the model compares the cumulative excess ^{210}Pb below that depth with the total unsupported ^{210}Pb in the core (Putyrskaya *et al.*, 2015).

$$t_x = \frac{1}{\lambda}.\ln\frac{C_r(0)}{C_r(x)} \quad = \frac{1}{\lambda}\ln\frac{\int_0^\infty \rho_b(x)\,C(x)\,dx}{\int_x^\infty \rho_b(x)\,C(x)\,dx} \tag{2.4}$$

where,

$\rho_b\ (x)$ = Bulk density of sediment (g/cm^3)

$C_r\ (0)$ = Total cumulated excess ^{210}Pb in sediment core (Bq/cm^2)

$C_r\ (x)$ = Cumulated excess ^{210}Pb in sediment below depth x (Bq/cm^2)

C_x = Activity concentration of the unsupported ^{210}Pb at depth x (Bq/g)

The sedimentation rate from CRS model is calculated using Equation 2.5 as presented by (Putyrskaya *et al.*, 2015).

$$R_S \quad = \lambda\,.\ \frac{C_r\ (x)}{C\ (x)} \tag{2.5}$$

where,

$C_r\ (x)$ = Cumulated ^{210}Pb in the sediment below depth x (Bq/cm^2)

$C\ (x)$ = Activity concentration of excess ^{210}Pb at depth x (Bq/g)

The results of CRS model can be validated using an independent time marker usually ^{137}Cs. Once the CRS and ^{137}Cs results are combined, a composite chronological model referred to as Constant Rate of Supply-Piecewise (CRS-Pw) is used (Zhang *et al.*, 2017b). In using CRS-Pw model for age determination along the core depth, cumulative depth instead of actual depth is used to cater for sediment compaction (Putyrskaya *et al.*, 2015).

According to Putyrskaya *et al.* (2015) and Zalewska *et al.* (2015) excess ^{210}Pb is useful in determining the age of sediment deposited during the past 100 – 150 years period. Hence ^{210}Pb is useful in assigning dates to longer periods than ^{137}Cs which is limited to the 1950s where nuclear weapons were used. Further, ^{210}Pb is useful in areas where ^{137}Cs concentration is low especially in the southern hemisphere (Porto *et al.*, 2006; Shakhashiro and Mabit, 2011). The ^{210}Pbex is useful in investigating environmental changes and anthropogenic impacts on lakes and reservoirs (Yang *et al.*, 2014; Bhuiyan *et al.*, 2015). According to Dai *et al.* (2007); Yang and Turner (2013); Yusoff *et al.* (2015) and Zalewska *et al.* (2015) ^{210}Pb radionuclide aids in characterizing changes of sediment input into a lake or reservoir. The ^{210}Pb dating method is useful in understanding what has happened in the recent past which would be an insight of impacts of human activities. Hence, the dating is useful in apportioning sediment contaminants/pollutants to a potential source by relating the contamination changes with changes within the catchment.

2.5 Sediment Characterization

Geochronological and geochemical analysis of lake/reservoir sediment is useful in environmental management (Zalewska *et al.*, 2015). Sediments act as indicators of environmental change since they can inform on variations in sediment inputs and characteristics over time (Pilgrim *et al.*, 2015). In managing lakes and reservoirs, knowledge of sediment characteristics is paramount since the sediments have potential to strongly influence the chemistry of the water column through various diagenetic processes (Eilers & Gubala, 2003). In investigating lake/reservoirs quality, sediments core plays a significant role. The sediments give information on concentration of nutrients, heavy and trace metals. Sediment analysis aids in studying anthropogenic influences and available spatial temporal variation of contaminant concentration (Skordas *et al.,* 2015). The findings from these analyses inform the environmental managers on historical impact of anthropogenic activities on the lake which in turn helps to assess the effectiveness of conservation practices adopted within the lake's catchment. However, in Kenya, little has been done in characterizing lake sediments.

According to Sahli *et al.* (2017) and Yanhong *et al.* (2004) lake sediments hold a large volume of information on environmental characteristics, land use, and climatic changes in the lakes/reservoirs catchment. This information can be used to reconstruct the pollution history of a waterbody (Skordas *et al.*, 2015; Al-Mur *et al.*, 2017). Lake sediments aid in establishing the health status of aquatic ecosystems (Purushothaman *et al.*, 2012; Bing *et al.*, 2013). Thus, the analysis of sediment geochemistry is useful in assessing the possibility of toxic effects of heavy metal to waterbodies (Tarras-Wahlberg *et al.*, 2002; Seshan *et al.*, 2010; Ntakirutimana *et al.*, 2013). Though the metals are not biodegradable and have low solubility they easily interact with organic and inorganic matters and later settling to the water body bottom sediments (Al-Najjar *et al.*, 2011).

When geochemical changes are investigated along the sediment core depth, variation in contaminant loading with time can be assessed. This is usually the case when sedimentation rates are known (Tarras-Wahlberg *et al.*, 2002). Further, the study of sediment contamination along the core depth, also aids in studying the impacts of anthropogenic activities on aquatic system (Wang *et al.*, 2016; Zanjani-Jamshidi and Saeedi, 2017). Reliable, historical natural and anthropogenic metal accumulation over time is extracted from sediment core (Yusoff *et al.*, 2015; Al-Mur *et al.*, 2017; Sahli *et al.*, 2017; Zanjani-Jamshidi and Saeedi, 2017). Trace and heavy metals into the lakes /reservoirs are either from natural or anthropogenic sources (Seshan *et al.*, 2010; Rejomon *et al.*, 2016).

According to Bing *et al.* (2013), analysis of the lakes/reservoir's sediment, aids in establishing sources of pollutants within aquatic ecosystems. Some of the common sources of pollutants to aquatic ecosystem includes; surface water runoff, agricultural fertilizer, anthropogenic activities such as disposal of untreated industrial effluents and domestic sewage into the water bodies. This leads to an increase of nutrients, trace and heavy metals concentration in lake sediment (Ntakirutimana *et al.*, 2013; Al-Mur *et al.*, 2017). In depth information of nutrients, trace and heavy metals sources can be studied from a combination of sediment characteristics and statistical analysis such as multivariate analysis (Yusoff *et al.*, 2015).

2.5.1 Multivariate Analysis

Multivariate statistical analysis can handle large volumes of spatial and temporal data and aids in understanding their interrelationship. The multivariate statistical analysis extracts information on intercorrelated variables thus reducing on the need to assess individual variables. Such statistical techniques include; Principal Component Analysis (PCA), Factor Analysis (FA), Discriminate Analysis and Cluster Analysis (CA). The PCA is useful in assessing the sources of contaminants and it tends to reduce dimensionality of variables (Yuan *et al.*, 2014). The PCA converts the data into smaller set of independent variables thus aiding in interpreting variance in a large dataset of intercorrelated variables (Zhao *et al.*, 2012; Abdelhafez and Li, 2014). It is used to assess the sources of the variables by reducing the dimensionality and integrating majority of parameters into fewer factors. In using PCA, Principal Factors with eigen values greater than one are retained and the varimax loading per each group assessed. The resulting clusters aid in identifying homogeneous groups of variables that are classified to be from the same source (Yuan *et al.*, 2011).

Further, various statistical techniques including Cluster Analysis (CA), Principal Component Analysis (PCA), factor analysis (FA) and Discriminate Analysis (DA) are also useful in assessing temporal and spatial variation in water and sediment quality.

These statistical techniques are also exploited in identifying potential sources of contaminants either in water or sediment. For instance, Phung *et al.* (2015) successfully applied multivariate statistical techniques (CA, PCA, FA and DA) in evaluating spatial/temporal variations of surface water quality in Can Tho City, a Mekong Delta area of Vietnam. In assessing the spatial variability of river water quality and sources of pollution Khanday and Javed (2016); Sharma *et al.* (2015) and Zhou *et al.* (2007) applied the CA, PCA and correlation analyses. In various studies, the CA, PCA and correlation analysis have been found to be effective in determining underlying relationships between the water/sediment quality parameters and identify sources of pollution or contamination (Yuan *et al.*, 2011; Gupta *et al.*, 2014; Barakat *et al.*, 2016).

2.5.2 Possible Origins of Heavy Metals in Sediments

In sediment characterization, differentiating the elements originating from human activities and those from natural weathering is an important part of geochemical studies. To achieve this normalization is used where the metal is normalized to a textural or compositional characteristic of sediment. This aids in regional comparison of metal contents in the sediment and in determining Enrichment Factors (Bhuiyan *et al.*, 2015). The Aluminium (Al) and Iron (Fe) are widely used elements for normalization (Grunwald and Chen, 2006; Ghrefat *et al.*, 2011; Yusoff *et al.*, 2015; Al-Mur *et al.*, 2017). Usually the metal concentrations are normalized to an average shale value metal concentrations presented in Turekian and Wedepohl (1961).

The normalizing elements are considered conservative since they do not have significant anthropogenic sources and are resistant to chemical weathering thus do not actively participate in geochemical cycles (Tanentzap *et al.*, 2017). Hence, to investigate whether anthropogenic activities have had an impact on sediment quality various indices relying on normalizing elements are used. The main source of the normalizing metal should be the earth's crust. The element should compensate the influence of sediment characters on the elements concentration (Yuan *et al.*, 2011) The frequently used normalizing elements include; Aluminium (Al), Titanium (Ti), Lithium (Li) and Zirconium (Zr).

Indices such as geoaccumulation Index (I_{geo}), Enrichment Factors, (EFs), Contamination Factors (CFs), and Pollution Load Indices (PLIs) are useful in distinguishing elements from natural and anthropogenic sources (Addo *et al.*, 2011). These indices require selection of correct background values for various metals and elements under consideration (Liaghati *et al.*, 2003). For a reliable finding from the pollution indices such as CF, I_{geo} and EF, selection of correct representative background values for different elements plays a significant role (Liaghati *et al.*, 2003). In using EF, a reference metal is required where the metal used should be highly abundant in the sediment and not easily impacted on by anthropogenic activities. Proper selection of background values and use of these indices aids in understanding spatio-temporal variation of reservoir sedimentation. The variation of reservoir sedimentation over time can also be understood through modelling.

2.6 Modelling Erosion and Reservoir Sedimentation

Models used in erosion and nd sedimentation studies are classified based on model input, parameters and the extent of physical principles applied in the model. They are classified either lumped and distributed model based on the model parameters as a function of space and time and deterministic and stochastic models (Devia *et al.*, 2015). According to (Devia *et al.*, 2015) and (Gupta *et al.*, 2015), other classifications include whether the model is empirical (e.g.ANN), conceptual (e.g. HBV and Top model) or physically based (e.g. MIKESHE and SWAT model). To model erosion in a watershed, soil loss and spatial disaggregation for sediment delivery, various processes and scale of assessment should be considered (Karydas *et al.*, 2014). Some of the commonly used models are discussed in the following sub-sections.

2.6.1 Erosion Productivity Impact Calculator

Erosion Productivity Impact Calculator (EPIC) model is used to determine the effects of management strategies on soil and water through evaluating the effect of soil erosion on soil productivity (Yüksel *et al.*, 2008). The EPIC model is also used in hydrology, erosion and sedimentation, and weather simulation. The model is also used to assess pesticide status, plant growth, soil temperature different tillage operations and control of plant environment (Yüksel *et al.*, 2007). However, this model requires large amount of data especially on plant growth. Thus it is a major challenge to apply this model in Kenya due to the limited available data.

2.6.2 Universal Soil Loss Equation

Universal Soil Loss Equation (USLE) was developed by Wischmeir and Smith in 1978 and it is an empirically based, static, lumped model that is based on long-term expertise and many runoff plot data (Amore *et al.*, 2004). The model was based on statistical analysis of data collected on humid agricultural areas from 47 locations in 24 states of Eastern and Central region in USA (Yang and Sañudo-Wilhelmy, 1998). The USLE model considers parameters such as topography, soil erodibility, rainfall erosivity and management practice. The combination of the parameters is as given in Equation 2.6 and are considered to have a linear relationship.

$$A = R \times K \times LS \times C \times P \tag{2.6}$$

where,

A = Soil loss due to surface erosion (t/ha/yr)

R = Rainfall erosivity factor (MJ/mm/ha/yr)

K = Soil erodibility factor (t ha/MJ/mm/ha)

LS = Topographic factor (L = Slope length factor, S = Slope steepness factor in m/m)

C = Crop management factor

P = Erosion control practice factor

The model is useful in areas with limited available data (Kinnell, 2010) and it is applicable worldwide. On the other hand, Perovic' *et al.* (2013) reported that the model is limited in estimating sediment yield, transport, deposition, channel and gully erosion. In addition, the model is not spatially and temporally explicit. The prediction accuracy of USLE model is reduced when erosion from a single event is predicted. To improve accuracy on erosion assessment and the spatial, temporal limitation of USLE model, the model can be integrated with GIS and remote sensing for better results. For instance, Kefi *et al.* (2011) reported that USLE combined with GIS is useful to decision makers in establishing strategies of soil and water conservation. However, to improve on modelling of sediment delivery within a catchment, RUSLE, WEPP and SWAT models could be exploited (Alewell *et al.*, 2019). The new developments, has given impetus to the study of soil erosion and sedimentation in general

2.6.3 Revised Universal Soil Loss Equation

The Revised Universal Soil Loss Equation (RUSLE) was developed by Renard *et al.* (1997) and is an improved version of USLE. It is also an empirically based, static and lumped model which is useful in water erosion. Renard *et al.* (2011) reported that RUSLE model is based on USLE model with revised K and R factors and it acts as a transition from empirical to physically based models. The RUSLE is useful in estimating the annual soil loss per unit area. The data required include; soil type, climate, topography and land cover types. It has six parameters; rainfall erosivity, soil erodibility, conservation management practice, slope length, slope steepness and vegetation cover.

According to Hrissanthou (2011), RUSLE application to large basins is limited in the sense that, the model does not consider sediment deposition nor is it spatially/temporally explicit. RUSLE model is not data intensive and can be used in a GIS environment where locations of erosion are isolated on a cell by cell basis. The model has been combined with GIS to improve on its spatial distribution, where satellite images are used in derivation of vegetation cover (C) and conservation practices (P). In addition, the Digital elevation Model (DEM) is used in determining topographic factor (LS) while the Soil Erodibility factor (K) is derived from soil map. Ayalew (2014) reported that when RUSLE and GIS are combined, it is possible to estimate spatial distribution of soil loss. Though the model has been used in soil erosion studies, it is not suitable in simulating sediment deposition in reservoirs.

2.6.4 GeoWEPP model

This is a physically based numerical process model that is used in soil erosion prediction. It was developed by Renschler and Lee (2005). It has a capability of simulating erosion from both hill slope version and a watershed version where hill slopes are combined with channels and impoundment elements (Flanagan *et al.*, 2013). The GeoWEPP integrates the Water Erosion Prediction Project (WEPP), Topography PArameteriZation (TOPAZ) and ArcGIS in sediment yield prediction (García-Lorenzo and Conesa-García, 2009). According to Ramsankaran *et al.* (2009) Water Erosion Prediction Project (WEPP) model is process based and used to estimate sediment yield and soil erosion.

The WEPP model considers the soil type, climate, ground cover percentage and topographic condition. It simulates erosion spatially and temporally (daily, monthly or annually). The WEPP model can be applied to either hill slope profiles (1 to 200 m in length) or small watersheds (up to about 260 ha) that comprise multiple hill slopes, channels, and impoundments (Renschler and Lee, 2005). For a hill slope profile model simulation, the minimum input requirements to the model are climate, slope, soil, and cropping/management input files (Dehvari, 2014). In watershed simulations, additional inputs needed are a watershed structure file, channel parameter files (for each channel), and impoundment parameter files (for each impoundment, if any). All inputs to WEPP are in ASCII text files, which makes it relatively easy for the creation of user interfaces by either the model developers or by other user groups (Zhang *et al.*, 2015).

The input files to GeoWEPP are generated within WEPP where topographic data from DEM is parameterized by TOPAZ. The input data include; DEM, climate data, soils data and land use map (Yüksel *et al.*, 2008). The data is mostly projected into UTM and converted to ASCII raster file. The DEM is used to delineate channel network, hill slope and in generating slope length. The sub - watersheds are then established. From the input data, four input files namely; climate, slope, soil and management files are generated for GeoWEPP model (Zhang *et al.*, 2015). These are used to describe the hill slope, geometry, meteorological characteristics, soil properties and ground cover respectively. On the other hand, GIS function is used in generating watershed outputs.

According to Yüksel *et al.* (2007), CLIGEN model which is a stochastic weather generation model is used in WEPP model to produce climate files. The climate file comprises daily precipitation, temperature, solar radiation and wind speed. This model considers spatial and temporal distribution of erosion and deposition (Landi *et al.*, 2011). Baigorria and Romero (2007) reported that the model is also useful in assessing the impact of different management practices in reduction of erosion and sedimentation. Further, the model is useful in establishing areas within the catchment that should be considered for prioritizing conservation practices (Flanagan *et al.*, 2013).

2.6.5 Water and Tillage Erosion Model/ Sediment Delivery Model

Water and Tillage Erosion Model/ Sediment Delivery Model (WATEM/SEDEM) is an empirically, spatially distributed model which was developed by Van Oost *et al.* (2000) and Van Rompaey *et al.* (2001). The model handles data with finite raster element at a catchment scale and it can estimate the catchment sediment yield. It is useful in determining sediment transport and delivery and it has high prediction power. The eroded sediment is routed through the basin to a water body or it is re-deposited on the slopes based on transport capacity of overland flow per grid cell (Van Rompaey *et al.*, 2003). It considers sediment transport capacity to be proportional to potential rill erosion which caters for spatial variation during modelling process and uses RUSLE 2D in establishing soil loss (Shi *et al.*, 2012). In determining the sediment transport capacity, Equation 2.7 by Van Oost *et al.* (2000) is used.

$$Tc \quad = K_{Tc}\ R\ K\ (LS - 4.112\, Sg) \qquad (2.7)$$

where,

Tc = Transport Capacity (kg/m/yr)

K_{Tc} = Transport capacity coefficient (m)

R = Rainfall erosivity factor (MJ mm/m^2/h/yr)

K = Soil erodibility factor (kg h/MJ/mm)

LS = Topography factor

Sg = Slope gradient (m/m)

In use of RUSLE 2D for soil erosion estimation the RUSLE parameters are used as given in Equation 2.8:

$$E = LS_{2D} \times R \times K \times C \times P \qquad (2.8)$$

where,

E = Mean Annual Soil loss (kg/m^2/yr)

LS_{2D} = Slope length factor

Shi *et al.* (2012) reported that the model was easily adapted to environment outside Europe by calibrating the transport capacity coefficient. According to Ciampalini (2010), WATEM/SEDEM deals with geomorphological modelling since it considers main hydrological and sediment erosion transport rules. It is also useful in identifying impact of different management activities in controlling erosion.

The input to the model includes; DEM, land use and soil map. It is considered that in each pixel, the amount of sediment input is added to soil erosion in that cell. Where transport capacity is higher than total sediment input and sediment production, the sediment is routed further down slope. On the other hand, if the transport capacity is lower than sediment input then net sediment deposition occurs. The model has an output of a pixel map which represents the amount of net erosion and net deposition at each pixel. The model focuses on spatial variability and is useful in estimating spatial pattern of soil loss, sediment transport across land units and sediment deposition. The model is mainly used in small catchments unlike the Soil Water Assessment Tool (SWAT) which is applicable in large catchments like that of Lake Naivasha.

2.6.6 Soil Water Assessment Tool

Soil Water Assessment Tool (SWAT) is a continuous simulation model developed by the USDA-Agricultural Research Service and Texas A&M University (https://swat.tamu.edu) and the model is available for public use. It is a robust model in studying agricultural dominated watershed (Sang *et al.*, 2015). SWAT is a physically based distributed and continuous time and semi-distributed model that is useful in predicting the impact of land management practices on water, sedimentation, erosion, weather and agricultural chemical yields from complex watersheds (Arnold *et al.*, 1998; Moriasi *et al.*, 2011).

The model simulates various climatic and hydrologic processes and land management operations at basin level (Arnold *et al.*, 2012). In SWAT, the hydrological cycle is based on water balance equation (Equation 2.9) as reported by Arnold *et al.* (2012).

$$S_{wt} = S_{wo} + \sum_{i=1}^{t}[R_{day} - Q_{Surf} - E_a - W_{Seep} - Q_{gw}] \quad (2.9)$$

where,

S_{wt} = Final soil water content (mm)

S_{wo} = Initial soil water content on day (mm)

t = Time (days)

R_{day} = Amount of precipitation on day (mm)

Q_{Surf} = Amount of surface runoff on day (mm)

Q_g = Amount of return flow on day (mm)

E = Amount of evapotranspiration on day (mm)

W_{Seep} = Amount of water entering vadose zone from soil profile on day (mm)

On the other hand, erosion and sediment yield is estimated from each Hydrologic Response Unit (HRU) using Modified Universal Soil Loss Equation (MUSLE). MUSLE uses amount of runoff in simulating erosion and sediment yield while USLE uses the rainfall indicator of erosive energy. The sedimentation equation is given in Equation 2.10 adopted from Arnold *et al.* (2012).

$$Sed = 11.8 \times \left(Q_{Surf} \times q_{Peak} \times Area_{HRU}\right)^{0.56} \times K \times C \times P \times LS \times CFRG \qquad (2.10)$$

where,

Sed = Sediment yield on a given day (metric tons)

Q_{Surf} = Surface runoff volume (mm H_2O/ha)

q_{peak} = Peak runoff rate (m^3/s)

K_{USLE} = Soil Erodibility factor (0.013)

C_{USLE} = Cover and management factor

P_{USLE} = Support practice factor

$CFRG$ = Coarse Fragment factor

LS = Topographic factor

$Area_{HRU}$ = Area of HRU (ha)

The MUSLE equation accounts for antecedent soil moisture hence it can assess sediments from a single storm (Blanco and Lal, 2008; Neitsch *et al.*, 2011).

An interface of SWAT model to ArcGIS exists and it is referred to as ArcSWAT. The ArcSWAT allows GIS data to be easily formatted for use in the model. The main steps in ArcSWAT include; watershed delineation, Hydrologic Response Unit (HRU) analysis, weather data definition, SWAT simulation, sensitivity analysis and calibration. The SWAT model is robust in modelling watershed that have many sub watersheds.

In the model, routing of sediment transportation consists of deposition and degradation process. Deposition and degradation within a channel reach affect sediment transportation process. Hence, the deposition and degradation operating within a channel reach is computed using constant channel dimensions for the entire simulation or by downcutting and widening of the stream channel and where the channel dimensions are updated throughout the simulation (Neitsch *et al.*, 2002). Thus, sediment at the basin's outlet (S_{out}) is calculated using Equation 2.11 as after Neitsch *et al.* (2005).

$$S_{out} = S_{in} - S_d + D_t \qquad (2.11)$$

where,

S_{in} = Sediment entering the reach (tons)

S_d = Deposited sediment (tons)

Dt = Total degradation (tons)

To determine total degradation which is the sum of re-entrainment and bed degradation components, Equation 2.12 from Neitsch *et al.* (2005) is used.

$$D_t = (D_e + D_b)(1 - D_r) \qquad (2.12)$$

where,

D_e = Sediment re-entrained

D_b = Bed material degradation component

D_r = Sediment delivery ratio

Due to SWAT model distributed nature, it is useful in studying impacts of spatially varying characteristics to hydrological processes within a basin. The model has been extensively applied worldwide for hydrologic studies in relation to water and land resources management.

Various physical processes in a watershed can be simulated using SWAT. The processes include; surface runoff, evapotranspiration, infiltration, percolation in deep and shallow aquifers, lateral flow, channel routing and sediment. All the processes are simulated for the surface, percolation to shallow and deep aquifers, soils and vadose zone. Within the model, surface runoff, subsurface or lateral flow and return or base flow are considered to contribute to main channel. The SWAT model simulates processes within the watershed by dividing the river basin into sub basins. The GIS interface with SWAT enables subdivision of a basin to sub basins and Hydrologic Response Units (HRUs). Using a combination of soils, land use and slope the sub basins are subdivided into homogeneous HRUs (Moriasi *et al.*, 2011; Arnold *et al.*, 2012). From the HRUs, flow, sediment nutrients and pesticide loading are summed up and the resulting load are routed through channels, ponds and/or reservoir to watershed outlet (Neitsch *et al.*, 2011; Arnold *et al.*, 2012).

The input to the model is geospatial data which include; digital DEM, land use/land cover map, soil map and the hydro-meteorological data. The DEM is useful in defining watershed topography and is exploited in calculating sub basin parameters such as slope and definition of stream network. The soil characteristics and attributes are defined from the soil data while vegetation, and processes in lands and soils are provided by Land use data (Arnold *et al.*, 2012; Ghoraba, 2015). The model has the capability of integrating various hydro-meteorological parameters, management practices so as to simulate sediment yield (Sang *et al.*, 2015). In surface runoff estimation, the Soil Conservation Service method (USDA-SCS) also known as the Curve Number (CN) method and the Green and Ampt method is used. The Green and Ampt infiltration method uses hydraulic conductivity and metric potential of wetting front to estimate the infiltration rate. This method requires sub-daily precipitation data to estimate the infiltration rate. While using SCS curve number method, in surface runoff estimation, daily precipitation data is required. The soil type (permeability), land use and antecedent soil moisture conditions affect the curve number further impacting on amount of runoff generated. Equation 2.13 is used to estimate runoff depth in SCS curve number method (Arnold *et al.*, 1998; Arnold *et al.*, 2012).

$$Q_{surf} = \frac{\left(R_{day} - I_a\right)^2}{\left(R_{day} - I_a + S\right)} \tag{2.13}$$

where,

Q_{surf} = Daily accumulated surface runoff or rainfall excess (mm)

R_{day} = Rainfall depth for the day (mm)

I_a = Initial abstractions which includes surface storage, interception and infiltration prior to runoff (mm)

S = retention parameter (mm). The retention parameter depends on soil type, land cover management, slope, and antecedent moisture conditions. This is estimated using Equation 2.14

$$S = 25.4\left(\frac{1000}{CN} - 10\right) \tag{2.14}$$

where,

CN = Curve Number

Thus, Equation 2.13 is summarized to yield Equation 2.15 given as;

$$Q_{surf} = \frac{\left(R_{day} - 0.2S\right)^2}{\left(R_{day} + 0.8S\right)} \qquad (2.15)$$

$$R_{day} > I_a$$

The SWAT theoretical documentation by Neitsch *et al.* (2011) gives a detailed description of the various processes' computation within the model. Since SWAT model is robust in watersheds with varying soils, land use, and management conditions over long periods of time, it was found highly applicable in studying Lake Naivasha basin. According to Arnold *et al.* (1998) SWAT model can be used to assess the impact of vegetation growth, land use change and sedimentation on water quantity and quality making it an ideal model that was used in this study.

2.7 Impacts of Land Cover Change on Reservoir Sedimentation

Land use/land cover changes impact on biogeochemical and hydrological responses within a catchment. For instance, these changes can modify hydrology, local climate, precipitation, water quality, soil erosion, sedimentation and biological community structure (Pilgrim *et al.*, 2014). The impact of land use changes to transportation of sediment is mostly dominated by conversion of large forested and wood land areas to agricultural areas. Godwin *et al.* (2011) documented that Shiyang reservoir in China had been highly affected where 43% of wood land areas were converted into agricultural land. The authors reported that Ghana had experienced reservoir loss of 45% over a period of six years in Burekese catchment. This was attributed to human factors such as deforestation, population increase and poor catchment management education on communities. Increased peak flow during wet season with change in land use and land cover was reported to result in high turbid water indicating more sediments being carried into water bodies (Getachew and Melesse, 2012).

A study conducted by Loi (2010), on how land use change affects surface runoff and sediment yield in Tri An Reservoir, Dong Nai Watershed, Vietnam, showed that surface runoff increased by about 30% when 21% of forested area was converted to agricultural land. From the same study, sediment yield was found to increase by about 58.4%. Kimwaga *et al.* (2012) used SWAT model to quantify sediment loading into Lake Victoria with change in land use/cover of Simiyu catchment. The results from the model simulation and actual measured sediments were 98, 467 tons/year and 2,075,114 tons/year. This led to the conclusion that SWAT model underestimated sediment yield from the catchment. Further, Masinga reservoir in Kenya has been reported to have experienced a loss of about 10.1% of its storage capacity by 2011 over a period of 30 years. This loss was attributed to increased catchment activities such as increased farming on steep slopes of Masinga catchment (Bunyasi *et al.*, 2013). The Masinga dam catchment was also reported to have lost 62% of forest cover between the years 1976 and 2011. Also, Nzeve *et al.* (2014) reported that presence of heavy metals in Masinga dam sediments was an indicator that anthropogenic activities impacted on the aquatic ecosystem. However, the long-term changes in Lake Naivasha sedimentation due anthropogenic activities under different conservation practices is not well understood.

2.8 Impact of soil conservation practices on sediment yield

Implementation of soil conservation measures within a basin minimizes loss of fertile soil thus enhancing productivity of land in soil erosion prone areas (Napier and Cockerill, 2014). The conservation measures reduce erosion thus lowering sedimentation of waterbodies and aids in moisture retention within the farms. Most soil conservation technologies are either agronomic, vegetative, structural or management. Agronomic technology includes practices such as intercropping, contour farming and mulching while vegetative measures include tree planting, hedge barrier, grass or filter strips (Blanco and Lal, 2008). On the other hand, structural technology includes practices such as graded banks, stone bunds and bench terraces. The conservation measures classified under management technology comprise land-use change, rotational grazing and area.

Terraces are commonly practiced in Kenya. According to (WOCAT, 2007) fanya juu terraces first came into prominence in Kenya in the 1950s, but the rapid spread of the technology between 1970s and 1980s with the advent of the National Soil and Water Conservation Programme. The fanya juu terraces can be used in semi-arid and sub humid zones. In semi-arid zones they are usually constructed on the contour to hold rainfall where it falls while in sub-humid areas, they are graded in such a way that they discharge excess runoff (Critchley, 1991). The terraces are then stabilized with use of grass mostly the napier (Pennisetum purpureum) which serves as fodder and/ or with multipurpose trees (for instance Grevillea Robusta and citrus) which are planted on embankments (Herweg and Ludi, 1999).

CHAPTER THREE

MATERIALS AND METHODS

3.1 Study Area

The Study was conducted in Lake Naivasha and its basin. The Lake basin lies between latitude 0° 8’ 35” and 0° 54’ 53”S and Longitude 36° 04’ 43” and 36° 42’ 24” E having a catchment area of approximately 3400 km^2 (Stoof-Leichsenring *et al*., 2011). The lake’s surface area is approximately 145 km^2 which fluctuates from 100 – 180 km^2 (Becht *et al*., 2005). The Lake is located at an average elevation of 1,890 m above sea level (Becht and Harper, 2002). Surface inflows into the lake are from two perennial rivers namely Malewa and Gilgil together with an intermittent Karati River. Details are presented in Figure 3.1 which shows the Location of Lake Naivasha, its basin in Kenya, its two satellite lakes (Crescent and Oloiden) and the main inflow rivers.

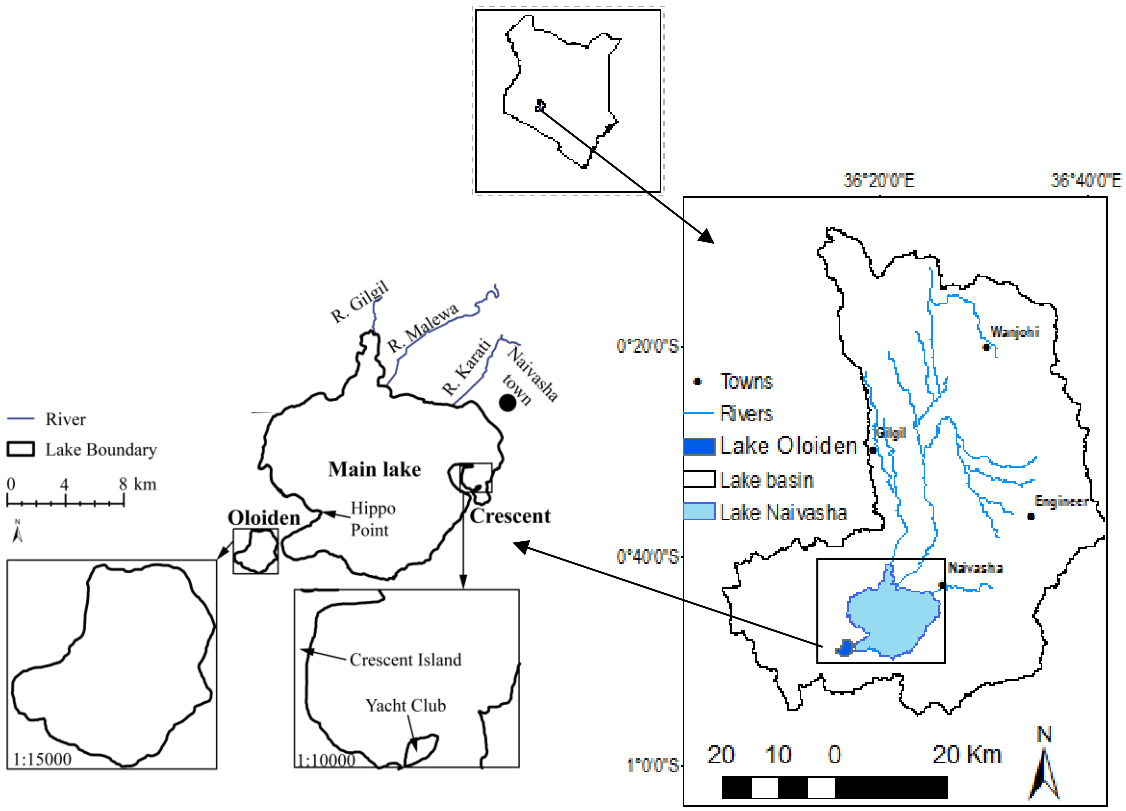


Figure 3.1: Map of Lake Naivasha basin in Kenya

According to Boar and Harper (2002) the Karati River flows into Lake Naivasha for about 100 days in a year. Lake Naivasha is a freshwater lake that has no surface outlet but its freshness results from the lakes interaction with ground water (Everard *et al.*, 2002; Bergner *et al.*, 2003). Hickley *et al.* (2004) and Kitaka *et al.* (2002) reported that the lake water balance mainly benefit from Malewa River which drains Kinangop Plateau. On the eastern, western and southern side of Lake Naivasha catchment, there are seasonal rivers that do not reach the lake, but they usually get underground. However, on the eastern side of the lake, Karati river flows into the Lake during the high rains while other rivers such as Marmonet from Mau Escarpment which recharges Ndabibi Plains end before they get into the lake (Becht and Harper, 2002).

The hydrogeology of Lake Naivasha basin can be classified into three zones namely; recharge, transit and discharge zone. The recharge zone is at the periphery of the basin and includes; east highlands of Nyandarua mountains and Kipipiri, Mau catchment to the west and Eburu in the North West. Drainage from Mau hills and Eburu infiltrates before reaching the lake (Becht, 2007). Lithosols and solentz soils exist around the southern shore of the lake (Bergner *et al.*, 2003). The soils were derived from volcanic ashes and other pyroclastic rocks formed during volcanic activities.

The climate in Lake Naivasha basin varies across locations. Around the lake area semi-arid conditions are experienced while on the upstream area, cooler and humid conditions exist. The bimodal type of rainfall is received with long rains being experienced between March and May, while short rains are received between October and November (Ndungu *et al.*, 2015). The annual rainfall ranges from about 650 mm to 1300 mm within the lake region and in Aberdares mountain forests, respectively (Odongo *et al.*, 2013). The average annual evaporation rate near the lake is 1,790 mm (Awange *et al.*, 2013). The average monthly temperature range is 15.9°C to 17.8°C with a mean monthly minimum temperature range of 6°C - 10°C. In addition, the mean monthly maximum temperature ranges from 26°C - 31°C (Kuhn *et al.*, 2012).

According to Hickley *et al*. (2002) intensive agriculture takes place within Lake Naivasha basin and has led to more areas being converted into farmlands even on steep slopes. The basin has undergone land use and land cover changes which has led to a decrease of pastureland (grass), wood land and sisal plantations which existed in the basin for the last 50 years (Harper *et al.*, 2011). Further, the need to increase agricultural land has led to excessive destruction of lake shore vegetation and forests (Kuhn *et al.*, 2012). One of the shore vegetation that has been majorly reduced is the Papyrus (*Cyperus papyrus*). According to Harper *et al*. (2011) papyrus currently occupies only 10% of the area it occupied in 1980’s. The reduction of papyrus which acts as a sediment filter could lead to increase of lake sedimentation. The effects of shore line development include increase in soil erosion, transportation and deposition of the sediments into the lake (Stoof-Leichsenring *et al*., 2011). The sediments may be highly contaminated with nutrients like phosphorous and other agro-chemicals from agricultural land which negatively affect the aquatic ecosystem.

3.2 Sedimentation Assessment in Lake Naivasha

The sedimentation of Lake Naivasha was analysed using bathymetric surveys, sediment coring chronological analysis and model simulation. The sedimentation status of Lake Naivasha was determined using Bathymetric Survey System (BSS) and sediment cores. The BSS has a multifrequency Acoustic Profiling System (APS), Vibe-core and a navigation system. The findings from bathymetric surveys; sediment coring and geochronological analysis were useful in establishing sedimentation status of the lake. The data was also useful in model calibration and simulation.

3.2.1 Bathymetric Survey

This involved determining the volume of water and sediments in the lake. The bathymetric survey of Lake Naivasha and Oloiden was conducted between 24th July to 16th October 2016. Prior to the survey of Lake Naivasha and Oloiden, the lake boundaries were digitized from Digital Globe images accessed in March 2016 from Google Earth. The delineated lake area was subdivided into different transects and tie lines that were used to guide the survey. The series of transects and tie lines were created as shapefiles using ArcGIS. The spacing of transects and tie lines were 50 m, 100 – 200 m and 2 km (Appendix 1) in the shallow, medium and deep (middle of lake) parts of the lake, respectively.

These spacings were guided by previous survey reported by Ase *et al.* (1986) which showed that there was less variation in bottom topography at the middle of the main lake. Further, the 100 m spacing was used for transects covering Lake Oloiden which is a satellite lake of Lake Naivasha. Since Crescent Island Lake is topographically complex, (Ase *et al.*, 1986) transect spacing of 50 m was followed. Furnans and Austin

(2008) together with Cross and Moore (2014) reported that closer transects spacing during bathymetric survey of waterbodies improve the confidence and accuracy of water volume estimated. To improve on data collection accuracy for the lakes studied, tie lines were created with an orientation of about 90° to transect lines. The tie lines used in Lake Naivasha were approximately 50 m, 200 m and 2 km while tie lines used in lake Oloiden had a 500 m spacing. The predetermined transects and tie lines allowed maximum coverage of the lakes during survey.

According to Sekellick *et al.* (2013) the tie lines provide independent measurements of depth and can also be used as a quality control check in sections where the tie lines and transect lines intersect. The use of transects and tie lines to ensure comprehensive coverage during bathymetric survey was adopted from Dunbar *et al.* (1999); Odhiambo and Boss (2004) and Sang *et al.* (2017) in various bathymetry survey studies. The boundaries, transects and the tie lines were projected into Universal Traverse Mercator (UTM) zone 37S. The boundaries; transects and tie lines; were then loaded in a fish finder (*Raymarine, Dragonfly 8 model). The fish finder had dual frequency, and, in this survey, it was used for navigation purposes while multifrequency Acoustic Profiling System (APS) was used in water and sediment depth measurement. The multifrequency APS had 12, 50 and 200 kHz frequencies which aided in simultaneous measurement of water depth and sediment thickness. The high frequency was reflected at top of sediments while low frequencies penetrated the sediments aiding in determination of sediment thickness (Dunbar *et al.*, 1999; Moriasi *et al.*, 2011).

The multifrequency APS was mounted on a motor driven dual Jon boat (Appendix 3) which was moving at a speed less than 6 km/h. The speed lower 6km/h was useful in avoiding cavitation and turbulence around the depth of transducer thus ensuring that the data collected was of high quality. The speed of survey was adopted from Sang *et al.* (2017) who had used a similar speed in a bathymetric survey of Ruiru reservoir, in Kenya. During the survey, the predetermined transects and tie lines pattern were maintained except when obstructions such as water hyacinth and papyrus, fishing nets, partially submerged trees, shallow waters and hippopotamus were encountered. This led to slight modifications of the navigation path to avoid the obstructions.

The Bathymetric Survey System (BSS) has an in-built navigation system, hence the data obtained during the survey includes; coordinates and corresponding water and sediment depth at each point. Sediment cores were collected and analysed, to assist in the interpretation of the acoustic data especially in assessing sediment thickness. Dunbar *et al.* (1999), Austin *et al.* (2014) and Solis *et al.* (2014) had also combined sediment coring and acoustic data when interpreting sediment survey data collected using a multifrequency APS.

3.2.2 Sediment Coring

After the bathymetry survey, six cores were collected at predetermined sites that were well distributed within the Lake (Appendix 1b). To get the actual point of coring within the lake, the acoustic data collected during the bathymetry survey was used. The collected acoustic data during bathymetric survey was useful in determining the depth of sediment core penetration. The sediment cores were collected in 3-inch diameter aluminium tubes. Core collection sites were carefully selected to have a complete sequence of sediment deposition within the lake. The closest core collected near the lake's delta was about 2 km away from the delta. This accounted for the high flow energy and the tractive processes that would lead to disturbance hence affecting the sediment deposition sequence (Morris and Fan, 1998) in the lake. The delta zones were not sampled since these regions were covered with papyrus.

Sediment core samples were collected using the vibe-coring device. The vibe-core device consists of a vibrating core head with weight rig, check valve, and core tube. The vibe-core was vertically lowered into the sediment sample until a compacted layer was reached where the core tube movement stopped or in some sites such as Crescent until the core tube length could allow. The core was then retrieved using a winch. After retrieval of the sediment core sample, a measuring rod was inserted at the top of tube to aid in determining the top of sediment in the core tube. The tube was then capped, stored and transported to the laboratory in an upright position. The procedure of retrieving and handling the core had previously been applied by Dunbar *et al.* (1999) and Solis *et al.* (2012).

Upon transportation to the laboratory, the cores were longitudinally split into two, then sliced into layers according to visually identified stratigraphic changes. The sediment cores collected were used to establish the depth of sediments deposited in the reservoir. It was useful in confirming and validating the results obtained from bathymetric surveys. The sediment cores were also used in investigating the stratigraphic changes denoting accumulation layer thickness, date of the layer together with heavy metals and nutrients in the sediment.

3.2.3 Sediment Core Processing

The sediment cores collected at each sampling site were split into two, where one sample was used for geochronological and bulk density analysis while the other one was used for geochemical analysis. From the retrieved sediment core, bulk density, chronological and geochemical samples were picked. The bulk density (ρ) was estimated using standard ASTM method (D 2937-04). Further, sediment chronology along the core depth was assessed using ^{137}Cs and ^{210}Pb radioisotopes. Since gamma counting is time consuming and expensive the number of samples analysed for radioactivity were based on stratigraphic changes along the core depth. According to Yang and Turner (2013) very close spacing leading to contiguous counting of core samples does not improve on dating.

3.2.4 Geochronological Sediment Survey

The sediment samples were oven dried at 50°C, they were then disaggregated using a rubber mallet and then passed through a 2 mm mesh sieve. Samples that passed through the 2 mm sieve, were homogenized and filled in Marinelli beakers. They were then weighed and sealed using an aluminium composite foil. To attain radioactive equilibrium, the sealed sample was stored for minimum of 21 days. This period was more than five times ^{222}Rn half-life. This procedure of sample processing was adopted from Wu *et al.* (2008); Yang and Turner (2013) and Putyrskaya *et al.* (2015). Their findings reported that sealing of the sediment samples in aluminium foils reduced uncertainty in dating.

After 21 days, radioisotopes activity was determined by gamma-spectrometry, using Broad Energy Germanium detectors (BEGe-5030) of Canberra. The single photon peak efficiencies were calculated using LabSOCs (Canberra GmBH, Rüsselsheim, Germany) calibration software. According to Putyrskaya *et al.* (2015) the software accounts for detector characteristics and the gamma rays self-absorption, both in the sample and container. The input into LabSOCs software, includes; the sample density, filling height in the beaker, dimensions and material of the beaker holding the sample.

To ensure correct measurement of radionuclide was achieved, background measurements of ^{222}Rn were determined prior to placement of the sample in the detector. Further, before each measurement was conducted, the detector housing was cleaned. Air volume within the detector was reduced by packing lead shielding Styrofoam plates which had dimensions of 30 cm diameter and 5 cm thickness.

After placing the samples into the detector, ^{40}K, ^{210}Pb, ^{214}Pb, ^{214}Bi and ^{137}Cs radio activity was determined. The radioactivity was determined after a count of 86,000s was attained. The ^{40}K, ^{137}Cs and total ^{210}Pb activities were measured at different channels of 1461 keV, 662 keV and 46.5 keV, respectively. In addition, ^{214}Pb and ^{214}Bi which are useful in calculation of excess ^{210}Pb from total ^{210}Pb (Sikorski and Bluszcz, 2003) were detected at 352 keV and 609 keV, respectively. From the detected activities, the dates were assigned on excess ^{210}Pb using Constant Rate of Supply-Piece wise (CRS-Pw) model (Equations 2.4 and 2.5) where ^{137}Cs was used as the independent time marker.

However, dating six cores was prohibited by associated cost, thus only two cores from Lake Naivasha were dated. Through correlation of synchronous layers, dates were assigned to cores that had not been radiometrically dated. The correlation between cores was carried out by identifying synchronous horizons along the sediment cores as described by Austin *et al*. (2014) and Shotbolt *et al.* (2005). With assigning of dates on sediment core, layers corresponding to the past 20 and 50 years were identified, and this aided in postprocessing of sediment data from bathymetric survey.

3.2.5 Post Processing of Bathymetric Data

During post survey processing of data, acoustic images were traced along a profile using a post processing and editing program, Depth Pic (Specialty Devices, Inc., Wyle, Texas). In Depth pic the multifrequency APS composite images and cores were added and displayed. From the multifrequency APS composite images, various layers depending on sediment characteristics were identified. The results from dating of the sediment cores were then used to establish layers corresponding to the past 20 and 50 years.

In the trace selection option, three frequencies with a false colour coding of RGB (Red, Green and Blue) were selected and values on trace scale representing 200, 50 and 12 kHz were varied. This variation was undertaken until pixel colour display representing 20 years and 50 years layer were distinguished (Figure 3.2).

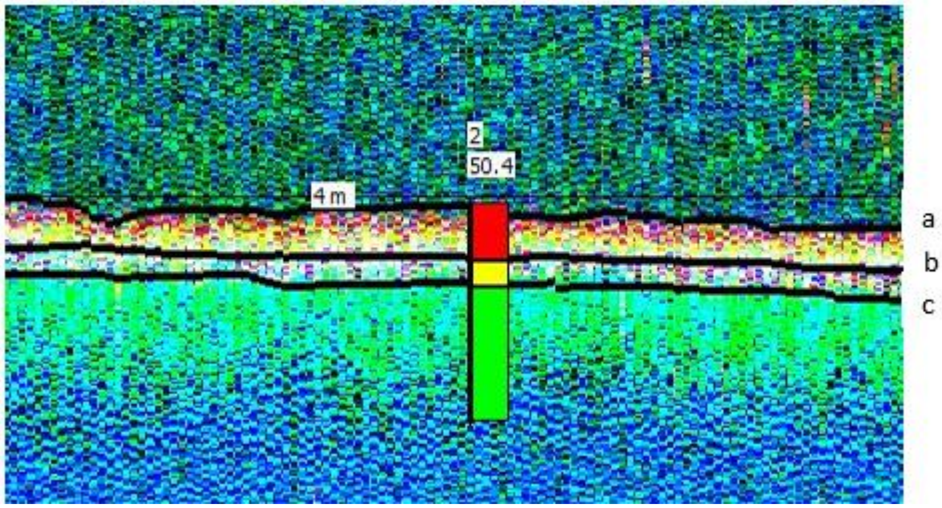


Figure 3.2: The multifrequency acoustic image showing the three distinct sediment layers corresponding to (a) surface, (b) 20 and (c) 50 years (from surface of sediment) together with a representation of sediment core

The identified layers corresponding to the past 20 and 50 years on the cores, were traced for all acoustic images collected as described by Dunbar *et al.* (1999). The tracing of layers in acoustic images aided in extracting XYZ (Latitude, Longitude, sediment and Water depth) values for all the points surveyed. The XYZ data were then imported into Surfer 14, Golden software (Golden Software, Inc. Golden, CO, USA) worksheet and saved as surfer data file. The duplicates and any outliers caused by poor response of GPS were removed during data cleaning. In processing bathymetric data Yesuf *et al.* (2013) and Sang *et al.* (2017) had used a similar procedure in data cleaning. A grid file from the point data was then created using ordinary Kriging interpolation method. According to Dost and Mannaerts (2008) and Aykut *et al.* (2013) ordinary Kriging interpolation method creates smoother contours than those generated from Triangulated Irregular Network (TIN) model. Sediment volume corresponding to a given period, water volume and areas were then calculated in Surfer 14. From sediment volume, Volumetric Lake Sedimentation Rate (LaSR) corresponding to 0-20, 20-50, and 0-50 years period was determined using Equation 3.1.

$$LaSR = \frac{SV}{Yrs} \tag{3.1}$$

where,

LaSR = Volumetric Lake Sedimentation Rate (m^3/yr)

SV = Sediment Volume (m^3)

Yrs = Number of years under consideration (20, 30, 50 years)

In addition, annual sediment yield was calculated using Equation 3.2 as described by Sekellick *et al.* (2013) and Tamene *et al.* (2006).

$$SY = 100\ \frac{SV \times dBD}{TE \times Yrs} \tag{3.2}$$

where,

SY = Sediment Yield (t/yr)

dBD = dry Bulk Density (t/m^3)

TE = Trap Efficiency (%)

Since Lake Naivasha has no surface outlet the TE was assumed as 100%, thus Equation 3.2 was summarized to the form presented in Equation 3.3.

$$SY = \frac{SV \times dBD}{Yrs} \tag{3.3}$$

The dBD for various layers was determined from the sediment cores collected. This was determined using standard ASTM method (D 2937-04). Further, Equation 3.4, adopted from Tamene *et al.* (2006) was used in determining the area Specific Sediment Yield for Lake Naivasha basin.

$$SSY = \frac{SY}{A} \tag{3.4}$$

where,

SSY = Specific Sediment Yield (t/km^2/yr)

A = Catchment area (km^2)

3.2.6 Generating Depth-Area-Volume Relationship of Lake Naivasha

The results on water volume aided in generating, water contours and Depth-Area-Volume (DAV) relationships. For contour presentation, the generated contours were then cartographically edited following a procedure described by Yesuf *et al.* (2013). On the other hand, raw contours were used to generate Depth-Area-Volume relationship. In calculating volume and area from the grid file, the inbuilt mathematical functions in Surfer software were used. This was achieved by calculating volume and surface area corresponding to various water depth contours at 1.0 m depth interval. The contours used ranged from Z= 0 m to Z= 17 m where 0 depicted the surface of the lake.

To present the variations between lake volumes, surface areas and various water depths, the established volumes and areas values were plotted against the depth contours resulting to a Depth-Area-Volume relationship. The topography of the lake considering water depth and sediment thickness were established in Surfer by using profiles that captured various sections of the lake. Sediment thickness was also confirmed using the sediment core that had been collected at various sites. The sediment core collected were also investigated for heavy metal contamination and the possible sources of sediment investigated.

3.3 Investigating Heavy Metals, Nutrients and Sediment Source

From six (6) sediment cores collected from Lake Naivasha, samples for heavy metals and nutrients were carefully picked from the cores. Sediments samples near the wall of the core tube were discarded to avoid cross contamination. In ensuring the analysed sample was free from core walls contamination, Wu *et al.* (2008) had followed a similar procedure of sampling out from core tube. After sample collection, geochemical analysis on sediment was undertaken.

3.3.1 Geochemical Analysis

The sediment samples were prepared, digested and analysed for heavy metals using Inductively Coupled Plasma Optical Emission Spectrometer (ICP OES). For analytical quality control; standard reference materials (GBW 07402 and GBW 07404), reagents blanks and randomly selected sample replicates were analysed. The blanks did not indicate any contamination and the standards confirmed the accuracy of analysis in ICP OES. To avoid possible contamination, the glassware used was acid washed by soaking them in Nitric acid-HNO_3 (10%) for at least 24 hours and rinsed repeatedly with distilled water. In preparation, the sediment samples were air dried and, in some cases, oven dried at 40°C. The sediment samples were crushed and then pre-sieved (2 mm) to eliminate gravel, shells and roots. To obtain homogenized fine powder, the sample was ground using a pestle and porcelain mortar.

From the ground sample, 0.25 g material was added into Teflon lined high pressure tubes which were used for heavy metal digestion. Digestion of heavy metal was achieved by using Aqua regia (HCl + HNO_3, volume ratio of 3:1). The samples and Aqua regia in Teflon tubes were mixed, placed in a microwave (Berghof MWS-3

Digester), and digested for about 15 minutes at 200°C. After the 15 minutes, the solution was left standing to cool, filtered and then diluted to 50 ml using distilled water. The procedure followed for microwave digestion of sediment samples was from EPA method #3051 (Appendix 2).

Following digestion of the sediment sample, analysis of heavy metals (Al, As, Cr, Cu, Fe, Mn, Ni, Pb and Zn) was performed using ICP OES (ICP OES Avio 200). A similar procedure of digesting and analysing sediments for heavy metal concentration had been followed by Hseu *et al.* (2002) and Islam *et al.* (2015). From the results of heavy metal concentration, statistical analysis was performed to aid in identifying possible sources of heavy metals in the sediment.

3.3.2 Identifying Potential Sediment Sources in Lake Naivasha

Potential sources of heavy metals in sediment samples were investigated using multivariate statistical techniques. The techniques used were Pearson correlation and Principal Component Analysis (PCA) where Analyse-it 4.92 extension in Excel software was used for analyses. Pearson correlation analysis aided in identifying relationship between heavy metals in sediment. Values above 0.65 were considered to signify strong correlation while those below 0.4 indicated that there was weak correlation between the heavy metals. A similar classification criteria had previously been used by Lim *et al.* (2013), Ghandour *et al.* (2014) and Bhuiyan *et al.* (2015); while investigating the sources of various heavy metals in water and sediments.

To group heavy metals that could be from same source, PCA was performed on the heavy metal concentration in sediments of different sites within the lake. The eigen vectors in PCA were used to determine the sites that had common pollution sources. In the analysis, Principal Components (PC) which had eigen values greater than one, were relevant in heavy metal classification (Yang *et al.*, 2014; Yuan *et al.*, 2014; Venkatramanan *et al.*, 2015). To consider heavy metals contribution to a given group, factor loadings to a component were used. The components that were found to have a factor loading >0.6, 0.4 - 0.6 and 0.3 - 0.4 were classified to be strongly, moderately or weakly associated with elements in that class, respectively. The classification criteria was adopted from Wu *et al.* (2014). The resulting clusters from the different components aided in establishing the groups of elements that may be classified as homogeneous. Yuan *et al.* (2011) had also applied Pearson correlation and PCA in assessing sources of contaminants.

3.3.3 Assessment of Sediment Contamination with Heavy Metal

Indices such as geo-accumulation index (I_{geo}), Contamination Factor (CF) and Enrichment Factor (EF) were used to assess the degree of sediment contamination with heavy metal. The indices use background values for various metals in the region. Since no studies have been conducted in investigating background values of metals within the region, global average for shale values reported by (Turekian and Wedepohl, 1961) were used. The CF index was used in determining the number of times heavy metal concentration in the sediment exceeded the background concentration. In calculating the CF, Equation 3.5 presented by El-Sayed *et al.* (2015) and Wang *et al.* (2016) was used. The CF results were then classified into grades based on groups reported by El-Sayed *et al.* (2015) and Al-Mur *et al.* (2017)

$$CF = \frac{C_{metal}}{C_{background}} \tag{3.5}$$

where,

C_{metal} = metal concentration in sediment (mg/kg)

$C_{background}$ = average background value of the metal in sediment (mg/kg).

Further, I_{geo} was used in assessing the extent of anthropogenic input in sediment. This was performed using Equation 3.6 as presented in Seshan *et al.* (2010); Sabo *et al.* (2013); Shafie *et al.* (2013); Gupta *et al.* (2014); Bhuiyan *et al.* (2015) and Elkady *et al.* (2015).

$$I_{geo} = \log_2\left(\frac{C_n}{1.5\,B_n}\right) \tag{3.6}$$

where,

I_{geo} = geoaccumulation index

C_n = measured concentration of specific heavy metal (mg/kg)

B_n = geo-chemical background of the heavy metal in consideration and (mg/kg)

1.5 = correction factor due to geogenic influences.

On the other hand, EF (Equation 3.7) is useful in regional comparison of heavy metals concentration in the background and sediment samples.

$$EF = \frac{\left(\frac{C_m}{C_{Al}}\right) sediment}{\left(\frac{C_m}{C_{Al}}\right) background} \tag{3.7}$$

where,

$\left(\frac{C_m}{C_{Al}}\right) sediment$ = the ratio of metal and Al concentrations in the sediment sample

$\left(\frac{C_m}{C_{Al}}\right) background$ = the ratio of metal and Al concentrations for the background values.

It is also useful in investigating human influence on heavy metal concentrations in the sediment (El-Sayed *et al.*, 2015; Al-Mur *et al.*, 2017; El-Amier *et al.*, 2017). In applying EF, selection of a normalizing metal which is not easily affected by anthropogenic activities is required. In this study, the procedure followed in normalization was adopted from Bhuiyan *et al.* (2015) and Al-Mur *et al.* (2017) where Al was used since it is a conservative element. According to Hyun *et al.* (2007), Yuan *et al.* (2011) and Tahiri *et al.* (2016) Al has relative stability in sedimentation hence useful in differentiating natural and anthropogenic input sources for the investigated heavy metals. The calculated EF values were then classified as described by Ghrefat *et al.* (2011) and Bhuiyan *et al.* (2015). The EF values of various heavy metals is also useful in quantifying the impact of different conservation on the waterbody (Liaghati *et al.*, 2003; Zanjani-Jamshidi and Saeedi, 2017).

3.4 Catchment Prioritization and Best Management Practices

To identify critical areas for conservation prioritization, the Soil Water Assessment Tool (SWAT) model was used. The model was set up, calibrated and validated then used to identify areas within Lake Naivasha basin which act as source of sediments. Further, simulation of various conservation practices was undertaken in a view of lowering sediment load into Lake Naivasha.

3.4.1 Setting Up the Model

Various datasets were used in the model set up, calibration and validation. Table 3.1 presents a summary of the data used in the SWAT model in this study. The DEM was useful in delineating Lake Naivasha basin and a threshold area of 6.1 km^2 was used in defining stream network and sub basins outlet. The soils within the basin were extracted from KENSORTER soils database. To generate the Land cover maps for Lake Naivasha basin, Landsat images for 1972, 1984, 2000 and 2015 were classified in ERDAS Imagine. The images were distributed over the study period with an intention of detecting any land cover changes in the basin. During the model set up, DEM, soils and land cover aided in creating sub basins and subsequently the HRUs.

Table 3. 1: Data used in SWAT model and their sources

Variables	Data	Source
Land use/land cover	Landsat MSS, TM, OLI images (1972, 1984, 2000, 2015) January season	https://earthexplorer.usgs.gov
Soil	Soil Terrain Database of Kenya (SOTER) Database Scale 1:1000000	ISRIC-WISE
Digital Elevation Model	Shuttle Radar Topography Mission (SRTM) 30 m resolution	https://earthexplorer.usgs.gov
Observed streamflow	Daily (1979-2016)	Water Resource Authority (WRA)
Temperature	Daily (1979-2016)	Kenya Meteorological Department, WRA and NasaPower
Precipitation	Daily (1979-2016)	Kenya Meteorological Department, WRA and Climate Hazard Group Infrared Precipitation (CHIRPS) data
Relative humidity, Windspeed and Solar radiation	1979-2014	Global weather data for SWAT
Bathymetric survey data	Lake Naivasha sedimentation survey	Sediment survey findings from current study

During sub basin delineation, river gauging stations that had observed data were selected to represent outlets of some sub basins. This was useful in the subsequent procedure of calibrating and validation of the model. The study area was subdivided into 71 sub basins that were useful in estimation of runoff and sediment yield. From the DEM, slopes were derived, and grouped into five classes at an interval of 10%. The classification of slopes was based on the recommendation in Kenya that prohibits practicing any agricultural activities on slopes greater than 20%. The Hydrologic Response Units (HRUs), were defined using the dominant land use, soils and slopes. Land use and soil data were reclassified, converted to shapefiles and were overlaid to aid in creation of HRUs. The sediment load estimation using SWAT model was undertaken for hydrological response unit. All the spatial data were then processed and saved as per the input requirement format of the model.

The weather data (daily rainfall, temperature, solar radiation, wind speed and relative humidity) of over 35 years period for the study area were input into the model weather database. The data quality was checked, and the missing data was estimated from the weather generator in SWAT model. Though SWAT model can simulate Potential Evapotranspiration (PET) using the Hargreaves, Priestley-Taylor and Penman-Monteith methods; Hargreaves method was chosen for this study. The choice was based on a study conducted by Kannan *et al.* (2007) which reported that a combination of CN method with Hargreaves methods of ET estimation yields reliable results from SWAT model. After the model was set up, it was calibrated for stream flow and sediment on annual basis due to lack of daily and monthly sediment data. Daggupati *et al.* (2010) and Tesfahunegn *et al.* (2013) got satisfactory results after setting SWAT model on annul basis due to lack of daily data.

3.4.2 Sensitivity Analysis, Calibration and Validation of SWAT Model

The most sensitive hydrologic and sediment transport parameters were determined using automatic sensitivity analysis. This aided in selection of parameters that were included in calibration process. After sensitive parameters were identified, those flow and sediment parameters were adjusted from SWAT initial estimates to fit the model output with observed flow and sediment data. During calibration, parameters affecting the flows were first changed. After the flow was calibrated and validated, the focus shifted to sediment calibration.

The streamflow and sediment were manually calibrated where the parameters were adjusted one at a time until the specified statistical calibration criterion were achieved. During calibration and validation, the available data was split into two datasets where one set was used for calibration and the other one for validation purposes. The model was calibrated and validated on annual basis where the data used for sediment calibration and validation was based on sedimentation survey and dating data. According to Tamene *et al.* (2006), the sedimentation data obtained from reservoir survey provides a more reliable indication of sediment loss and nutrient export from a catchment.

Every simulation was initialized by a warm-up period of 2 years. Calibration and validation periods were from 1981 to 1988, and 1989 to 1992, respectively. Considering the land use data of 1984 which represented 1979 – 1992 period, the warmup period for the model was chosen to be 1979 – 1980. To cater for land use changes over time the SWAT model was also set up for the periods between 1993 – 2004 and 2005 – 2015. The classified images of 1984, 2000 and 2015 were useful in uncovering the impacts of Land use/cover changes to sediment loading in Lake Naivasha. In this study, the land use/cover maps were used separately while all other SWAT inputs were similar. This ‘fixing-changing method’ meaning changing land use/cover maps and keeping other inputs constant have previously been used by Gashaw *et al.* (2018).

The modelling period selection considered was chosen to avoid rapid land use/cover change. The calibrated and validated parameters for the period between 1981-1992 were transferred to other study periods (1993-2004, 2005-2015) with adjustments being made where necessary. Some of the adjustments were on calibration changing parameters such as CN to reflect the change in land use. Statistical methods such as Nash Sutcliffe Efficiency (NSE), coefficient of determination (r^2) and Percent Bias (PBIAS) presented in Equation 3.8, 3.9 and 3.10, were used to determine the performance of the calibrated model.

$$NSE = 1 - \frac{\sum_{i=1}^{n}(O_i - P_i)^2}{\sum_{i=1}^{n}(O_i - \bar{O})^2} \tag{3.8}$$

where,

NSE = Nash-Sutcliffe Efficiency

O_i = Observed values

$\bar{O}$ = Observed mean values

P_i = Simulated values

The Nash Sutcliffe coefficient values range from ∞ to one where one is the optimal value. The r^2 which is defined as the squared value of the coefficient of correlation was calculated using Equation 3.9 as presented by Krause *et al.* (2005).

$$r^2 = \left[\frac{\sum_{i=1}^{n} (O_i - \bar{O})(P_i - \bar{P})}{\sqrt{\sum_{i=1}^{n} (O_i - \bar{O})^2} \sqrt{\sum_{i=1}^{n} (P_i - \bar{P})^2}} \right]^2 \tag{3.9}$$

where,

$\bar{P}$ = simulated mean values.

The r^2 estimates the combined dispersion against the single dispersion of observed and simulated series. The value of r^2 ranges between 0 and 1 where zero means no correlation and one denote that dispersion of the simulated is equal to that of the observed values

Further, PBIAS (Equation 3.10) which measures average tendency for simulated data to be smaller or larger than observed values was also used.

$$PBIAS = \left[\frac{\sum_{i=1}^{n} (O_i - P_i)}{\sum_{i=1}^{n} O_i} \right] \times 100 \tag{3.10}$$

where,

PBIAS = the deviation of data being evaluated (%)

The Positive PBIAS values indicate model underestimation while negative values indicate model overestimation bias (Gyamfi *et al.*, 2016). According to Moriasi *et al.* (2007) when both stream flows and sediment results yield NSE values ≥ 0.5, the model is considered to be satisfactorily calibrated. Further, considering PBIAS, the model performance is satisfactory when stream flows and sediments are within 25 and 55 % of observed flows and sediments, respectively (PBIAS ≤ ± 25% for stream flow and BIAS ≤ ± 55% for sediments).

In sediment calibration, sediment data measured at Lake Naivasha using Acoustic Profiling System (APS) was used. Hence, the cumulated sediment over the period of consideration was cross checked with those generated by the SWAT model. Licciardello *et al.* (2017) had also used cumulated sediment during SWAT calibration in a study that compared long term bathymetric measurements and SWAT estimations. Calibrated SWAT model is useful in estimating sediment yield from the basin. According to Yuan *et al.* (2009), for effective sediment yield reduction, it is important to first determine the source areas of the sediment yield where soil conservation works must focus on. Thus, in this study, SWAT model was used in identifying critical sediment source areas within Lake Naivasha basin and simulating catchment management scenarios.

3.4.3 Identifying Critical Areas for Sediment Yield in Lake Naivasha Basin

Sediment yield critical areas within Lake Naivasha basin, were identified and prioritized based on average annual sediment yield simulated using the SWAT model. The 2005 to 2015 period was considered as the base scenario (from 2015 Landsat imagery). Sediment yield from all sub basins was assessed, where sub basins with sediment yield ≥ 5 t/ha were identified as critical areas. According to Betrie *et al.* (2011) sub basins with sediment yield ≥ 5 t/ha are categorized as high sediment yielding sub basins thus they should be prioritized for conservation practices.

The identified critical sub basins were then arranged in descending order the first one representing the highest priority for conservation. This methodology had successfully been applied by Tripathi *et al.* (2003), Tesfahunegn *et al.* (2013) in assessing erosion hotspot for a sub-catchment management at Mai-Negus catchment in northern Ethiopia.

3.4.4 Catchment Management Interventions

This involved simulation of conservation practices that would reduce sediment yield from sub basins and sediment load into Lake Naivasha. The management practices were represented in SWAT model by modifying parameters that would reflect impacts of the practices on processes simulated by SWAT. The procedure followed is as presented by Arabi *et al.* (2008) and Betrie *et al.* (2011). The scenarios that were applied included; maintaining existing conditions for the period 2005-2015 as the base scenario; introduction of filter strips and terracing (Table 3.2). The width of filter strips was varied and their impacts on sediment load into Lake Naivasha assessed.

Table 3.2: Scenarios and parameters of focus in simulating conservation practices using SWAT model

S/N	Scenarios	Parameters on focus
1.	Base Scenario	2005 – 2015 period (Based on calibrated and validated model parameters
2.	Filter strips	FILTERW (Width adjusted from 0-27 m)
3.	Terracing	- SLSUBBSN (Dependent on slopes and determined from Horizontal interval) slope 0-10% = 24 m 10.1-20% = 20 m >20% = 10 m - USLE_P: varied between 0.1 – 0.18 Slope 0-10% = 0.1 10.1-20% = 0.14 >20% = 0.18 - CN: Subtract 6

In Scenario 1, filter strips were first applied on all HRUs that had been identified to be classified under high sediment yield category. Further, impact of filter strips in reduction of Lake Naivasha sedimentation was simulated in all HRUs that had agricultural land use. To assess impact of filter strips, the model parameter that was considered was the width of the filter strip (FILTERW). The FILTERW was varied between widths zero and 27 m. In varying the width, the Kenyan guidelines on riparian land (2 – 36 m) were considered as stipulated in Kenya Land Act, 2012. In addition, terracing was simulated for all HRU that had been classified to be under high sediment yield class and above, together with all HRUs that had agricultural activities. On the other hand, Scenario 2 (use of terrace) impacts were simulated in SWAT by modifying the CN, USLE support practice (USLE_P) and average slope length (SLSUBBSN). The value of SLSUBBSN was determined using the terrace guide presented in NRCS (2009). Horizontal Interval

(HI) that is useful in calculation of terrace spacing is equal to SLSUBBSN. The HI (m) was calculated using Equation 3.11 given in NRCS (2009).

$$HI = (xs + y) \times \frac{100}{s} \tag{3.11}$$

where,

x = Dimensionless variable ranging from 0.12 to 0.24. A value of 0.12 is used for high rainfall areas and that of 0.24 was used in low rainfall areas.

y = Dimensionless variable ranging between 0.3 to 1.2. According to (Arabi *et al.*, 2008; NRCS) 0.3 value is used for highly erodible soils with tillage systems that provide little or no residue cover. On the other hand, 1.2 is used on erosion resistant soils with tillage systems that leave a large amount of residue on the surface.

s = Slope of the HRU

For each class of slope used in this study, the corresponding SLSUBBSN was calculated and input in SWAT. A similar procedure of implementing filter strips and terraces had been followed by Mwangi (2011) and Gathagu *et al.* (2017).

Sediment reductions relative to the base scenario due to conservation practices was assessed using graphs and computation of percentage (%) reduction of sediment yield and sediment load. The percentage reduction was calculated using Equation 3.12 adopted from Verstraeten *et al.* (2006).

$$SR(\%) = 100\frac{S_B - S_C}{S_B} \tag{3.12}$$

where,

$SR(\%)$ = Sediment reduction (%)

S_B = Sediment from Base scenario

S_C = Sediment from conservation practices

CHAPTER FOUR

RESULTS AND DISCUSSION

4.1 Bathymetry Survey for Water and Sediment Depth

Results from 2016 bathymetric survey using multifrequency Acoustic Profiling System are presented here. Further, the Depth Area Volume relationships (DAVs) from the 2016 survey are presented. They are further compared with DAVs from the previous surveys of 1927 and 1983.

4.1.1 Water Depth in Lake Naivasha

The generated water depth contour map of Lake Naivasha and Lake Oloiden is presented in Figure 4.1. The zero m contour value corresponded to lake level at 1,889 m asl, as recorded from an official gauge at a yacht club (Figure 3.1). It was found that Hippo point in the Main Lake and Crescent Island Lake recorded maximum depth of 7 and 16.4 m, respectively. On the other hand, Lake Oloiden, a satellite lake to Lake Naivasha was found to have a maximum depth of 7 m. Comparing the 2016 water contour maps with those from 1927, 1983 and 1991 surveys, major differences were noted on 1927 map especially on the shape of Lake Oloiden map as presented by Ase *et al.* (1986). The 1927 survey had a poor coverage at the south-western part of the lake which could have led to a difference in Lake Oloiden shape. From the contour maps, it was observed that the presence of delta on the northern part of the lake was distinct in 1983, 1991 and 2016 survey while for year 1927 it was not distinct. The presence of delta from 1983 to 2016 contour maps could be associated with sediment loads deposition from the main inflows.

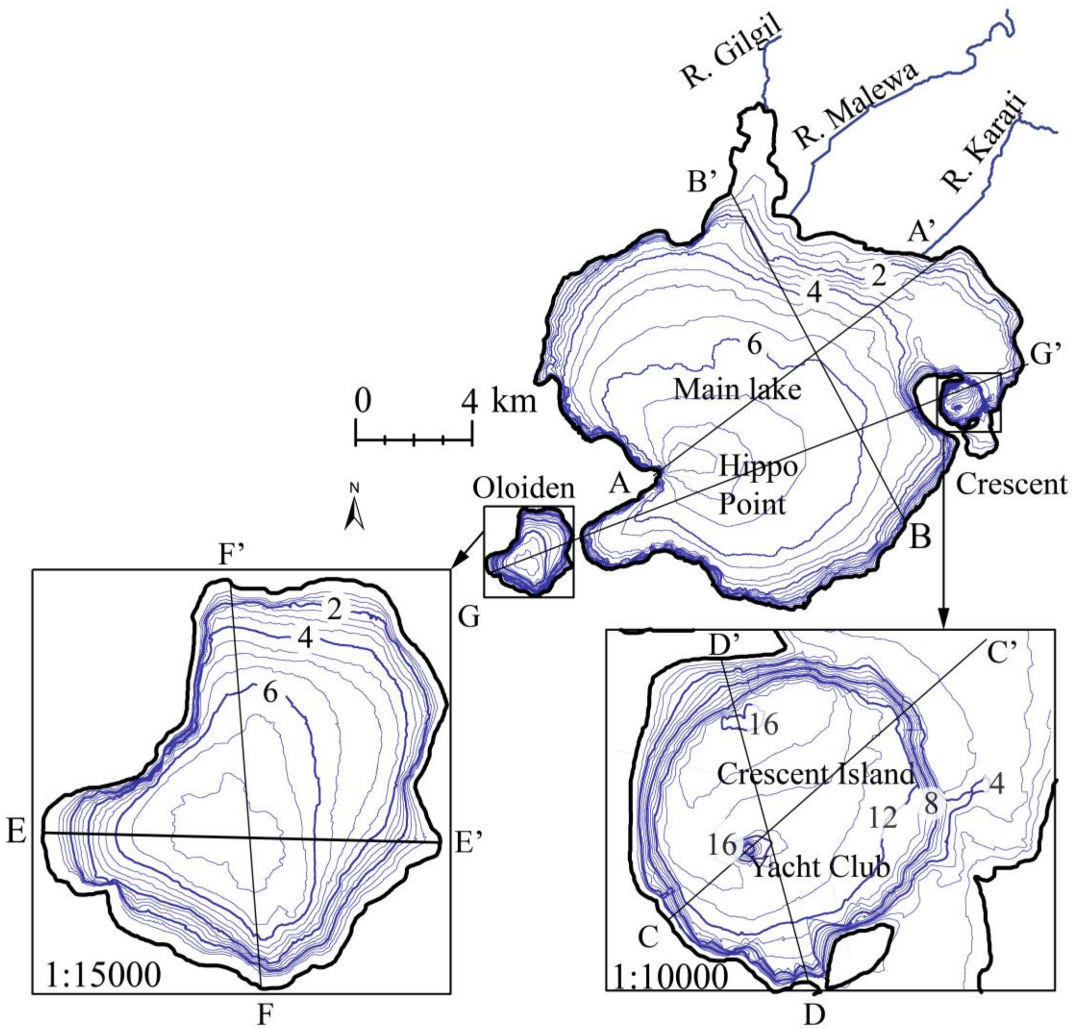
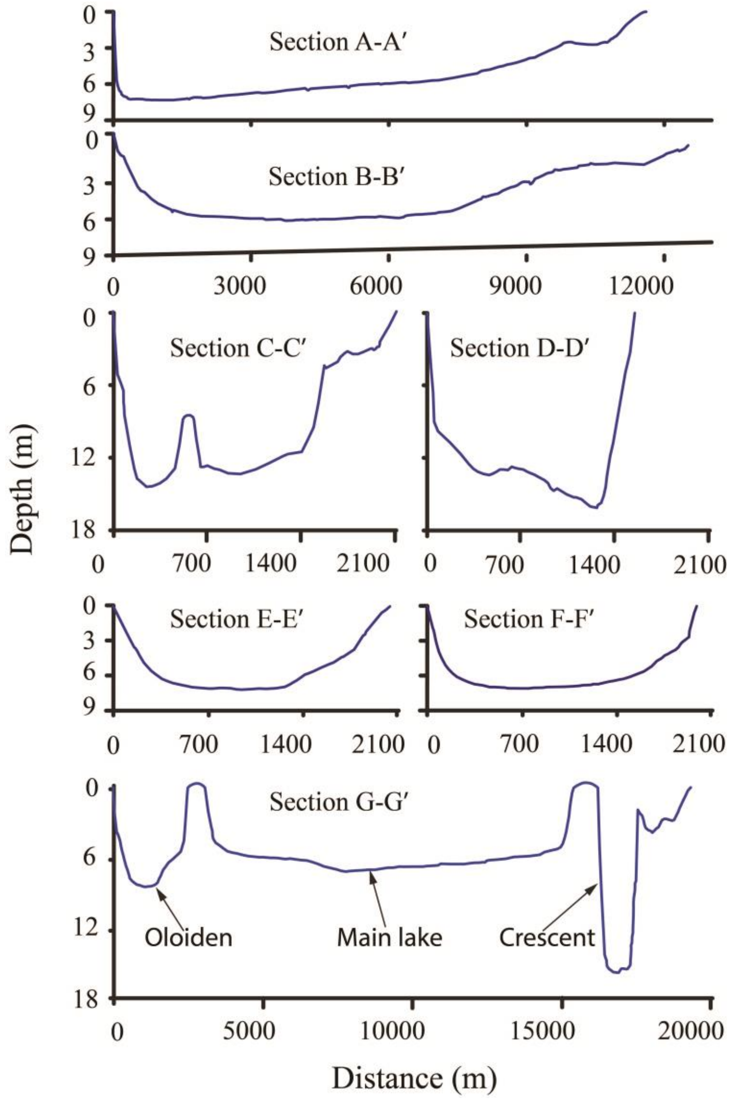


Figure 4.1: Profile tracks and water depth contours of Lake Naivasha and its Satellite Lake Oloiden based on 2016 bathymetric survey (The boundary corresponds to 1889 m asl)

From the 2016 survey, it was observed that the middle part of Lake Naivasha was generally flat. The findings from this study agreed with those reported previously by Thompson and Dodson (1963) and Ase *et al.* (1986). Further, from the profiles (Figure 4.2) gentler slopes were observed towards the northern parts of Lake Naivasha which covers the inflow area of the lake. This part could have gentler slope because of sediment deposition that takes place in the zone. A similar observation of inflow part of the lake having gentler slope had been made by Hassan *et al.* (2017) in a study on siltation rate for Dokan reservoir, Iraq.



- Section A-A’ and B-B’- Profiles from Main Lake
- Section C-C’ and D-D’ - Crescent Island Lake
- Section E-E’ and F-F’ -Lake Oloiden
- Section G-G’ - combined cross section running from Oloiden, Main Lake and Crescent Island

Figure 4.2: Cross section profiles of Main Lake, Crescent Island Lake, Lake Oloiden where 0 m = 1889 m asl

Between 1983 and 2016, the maximum depth in Main Lake (located at Hippo point), Crescent, and Oloiden lake were found to have reduced by 2.0 m, 0.6 m and 1.75 m, respectively. The change in maximum depth in various parts of Lake Naivasha could be due to sediment deposition or differences in adopted survey methodologies. Yesuf *et al.* (2013) reported that methodologies used during data collection and processing affects the results of bathymetric survey. Although water levels during 1983 and 2016 surveys were the same, the mean depth of Lake Naivasha in 2016 was found to be 0.23 m lower than that recorded from the 1983 survey as presented in Table 4.1.

Table 4. 1: Lakes characteristics for the previous years and 2016 bathymetric surveys

Parameters	Years			
	1927	**1983**	**1991**	**2016**
Volume (x 10^6 m^3)	870	900	-	722 († 748.2)
Surface area (km^2)	171	180	-	154.17 († 159.64)
Elevation (m)	1892	1889	1887.5	1889
Mean depth (m)	-	4.91	3.35	4.68
Maximum depth (m)				
- Main Lake (hippo point)	-	9	-	7
- Crescent	-	17	-	16.4
- Oloiden	-	9	-	7.25

† the volume and area represent Lake Naivasha and Lake Oloiden combined

It was observed that the mean depth recorded in 2016 survey was 1.33 m higher than that recorded in 1991 survey. In 1991 bathymetric survey, Lake Naivasha had a mean depth of 3.35 m (Hickley *et al.*, 2004). The water levels during 1991 survey was at 1887.5 m compared to 1889 m during 2016 which could explain the great difference in lakes mean depth between 2016 and 1991 surveys. However, reduction in mean water depth at a constant water level could be an indication that sedimentation has taken place in Lake Naivasha and its satellite lake over the years. This phenomenon would greatly affect the volume of the water held in the lake.

4.1.2 Lake Naivasha Water Volume

The volume of Main Lake and its satellite Lake Oloiden was found out to be 722 $x10^6$ m^3 and 26.2 $x10^6$ m^3 respectively. This corresponded to the lakes' surface areas of 154.17 $x10^6$ m^2 and 5.47 $x10^6$ m^2, respectively. Considering water level of 1889 m, the lake volumes from bathymetric surveys conducted in 1927, 1983 and 2016, are 730 x 10^6 m^3, 900 $x10^6$ m^3 and 748.2 x 10^6 m^3, respectively. Considering the lakes volume from 1927 and 2016 surveys (Lake Naivasha and Lake Oloiden combined), 18.2 x 10^6 m^3 increase in volume was observed though the water levels were the same. The difference in volumes could be due to poor transects coverage throughout the lake during previous surveys as reported by Thompson and Dodson (1963) and Ase *et al.* (1986). This could have resulted in increased errors on the established volumes. Further, reduced water volume in the lake at same surface water level of 1889 m, could indicate that there is a drop-in water depth due to sedimentation. According to Solis *et al.* (2012) comparison of lakes/reservoir volumes from multi-temporal studies aids in calculation of rates of volume loss especially where methodologies used from one bathymetric survey to another is the same.

4.1.3 Depth - Surface Area - Volume Relationships

Figures 4.3 a to d, present volumes and surface areas against depth at 1.0 m interval for Lake Naivasha and Lake Oloiden.

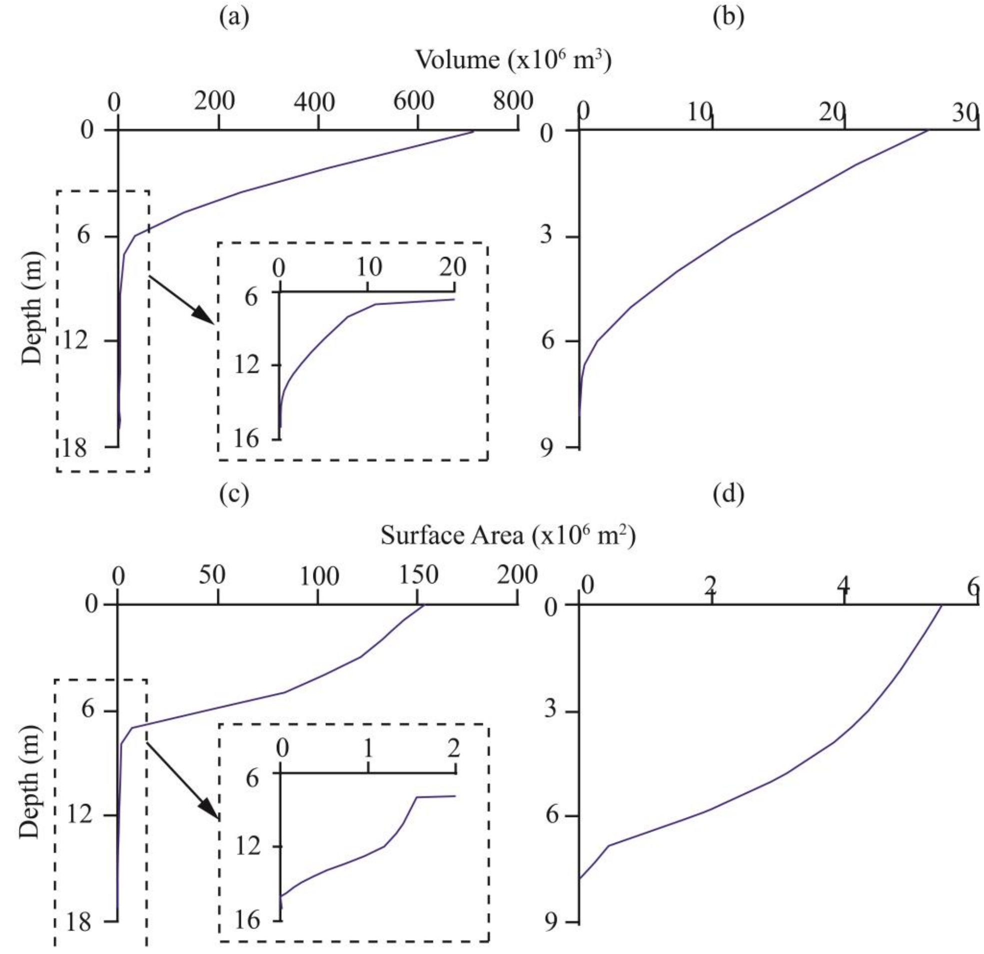


Figure 4.3: Depth, Surface area and Volume relationship (a and b representing depth volume relationship while c and d represent depth surface area relationship) of Lake Naivasha and Lake Oloiden, respectively. The 0 m = 1889 m asl

In comparing the 2016 Depth – Area - Volume curves of Lake Naivasha with those of previous studies (Figures 4.4 a and b), it was noted that the 1927 and 1983 curves did not consider depths beyond 10.0 m.

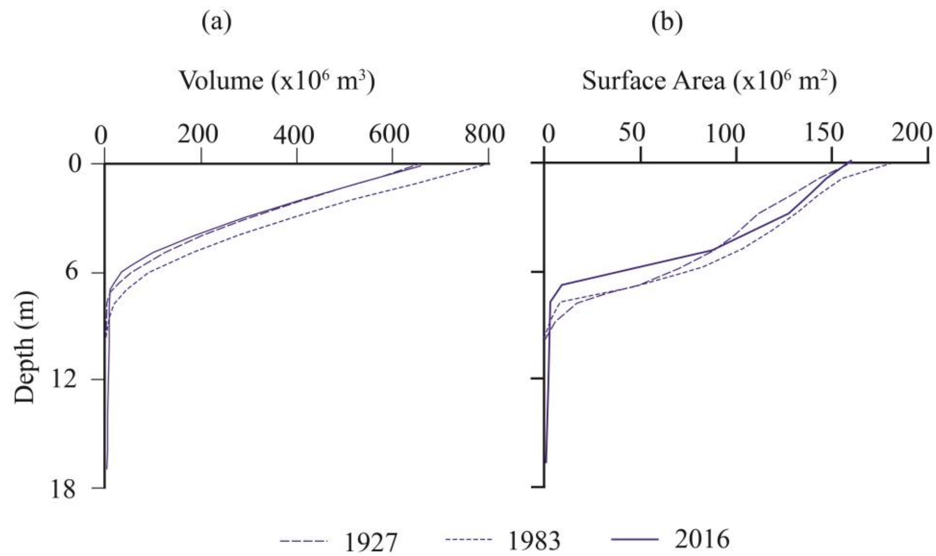


Figure 4.4: Depth, Surface Area and Volume relationship of Lake Naivasha for year 2016 and past surveys (1927 and 1983). The 0 m = 1889 m asl

The volume and area occupied by lake water between contours 5 and 10 m for period of 1927, 1983 and 2016, varied. The variation in curves could be due to the different methods and technological approaches used during the 1927, 1983 and 2016 surveys. For instance, Ase *et al.* (1986) had observed that theodolite equipment and few sections that were sounded during the 1983 bathymetric survey could have resulted to some errors and uncertainty in measurements. This could then affect the resulting Depth - Area - Volume relationship. On the other hand, the difference in resulting relationships could be attributed to autochthonous sedimentation taking place in Lake Naivasha. The Lake Naivasha Depth - Area - Volume relationship multi-temporal results had similar trends with those reported by Hassan *et al*. (2017) in a study conducted for Dokan Reservoir, Iraq. The changes observed on the Depth - Area relationship could also be because of presence of features that were unmapped in the previous surveys especially considering the wide spacing used for the previous surveys transects. McPherson *et al.* (2011) had also reported a change of Depth - Surface area relationship due to change of transect lines for a study conducted to determine Storage Capacity and Sedimentation in

Loch Lomond Reservoir, Santa Cruz County, California. The changes of Depth - Surface Area relationship of Loch Lomond reservoir was also attributed to variations of reservoir bed which could be due to sediment deposition. According to Rodrigues and Liebe (2013) the Depth - Surface Area - Volume relationship aids in establishing the storage capacity corresponding to a given depth and the sedimentation rate of reservoirs.

Availability and assessment of updated Depth - Area - Volume relationship would be very useful to the stakeholders and water managers in managing water withdrawals from Lake Naivasha. This is because Lake Naivasha serves both as a Ramsar site and it is also exploited for water supply and irrigation (Floriculture and horticulture) sector (Becht and Harper, 2002; Everard *et al.*, 2002; Hickley *et al.*, 2004; Mavuti and Harper, 2005; Mekonnen *et al.*, 2012). According to McAlister *et al.* (2013) and Yesuf *et al.* (2013), the Depth - Area - Volume relationships provide vital information on lakes and reservoirs that aid in their operation, prediction of sediment distribution and understanding of seasonal variations in storage capacities. The information from Depth - Area - Volume relationship becomes handy in allocating water abstractions (Ortt *et al.*, 2008).

Hassan *et al.* (2017) demonstrated the advantage of analysing Depth - Area - Volume relationship where the quantity of trapped sediments over a given period was computed by combining new Depth - Area - Volume relationship with the previous ones on the same figure. Issa *et al.* (2013b) established a reduction in reservoir capacity by comparing two subsequent surveys of Mosul dam, Iraq. The authors also observed that the shape of Depth - Area - Volume relationship changed with sediment deposition. However, Lake Naivasha water levels greatly fluctuates as shown in Appendix 3 (Hickley *et al.*, 2002; Bergner *et al.*, 2003; Bergner and Trauth, 2004; Becht *et al.*, 2005; Harper *et al.*, 2011; Yihdego and Becht, 2013). Thus, it is difficult to compare multi-temporal bathymetric surveys and correctly conclude that the capacity loss is due to sedimentation. Therefore, further analysis was conducted using multifrequency Acoustic

Profiling System, sediment coring and geochronological analysis to assess sedimentation of Lake Naivasha.

4.2 Sedimentation in Lake Naivasha

The physical and geo-chemical results from characterization of Lake Naivasha sediment are presented. Further, findings from radionuclide analysis, thickness, volume and distribution are also captured. Further, long-term sedimentation rates derived from bathymetric survey are given.

4.2.1 Physical Characteristics of Sediment Cores

It was found out that the water content in the sediment sample ranged from 23.2 to 97.6%. The samples with low and high-water content were mainly from mid-lake and Crescent Island Lake, respectively. According to Eilers and Gubala (2003) the samples with low water content usually indicate the presence of higher proportion of inorganic materials in sediments. In this study, it was observed that sediment samples with low water content registered high dry bulk density.

The dry bulk density of Lake Naivasha sediment ranged from 25 to 1513.9 kg/m^3 and an average density of 260.7 kg/m^3. The low-density sediment material was from the cores collected at Crescent Island Lake and corresponded with sediment depths of between 60 - 70 cm and 140 - 165 cm layer. This layer was identified by Verschuren (1999b) as having algal gyttja material that is characterized by having moisture content above 90%. The bulk density of sediments collected in Lake Naivasha was found to closely agree with those recorded for Lake Linganore, Frederick County, Maryland by Sekellick *et al.* (2013). The Lake Linganore sediment had bulk density of 228 and 1230 kg/m^3. According to Sekellick *et al.* (2013) most earth materials have dry bulk densities ranging between 1,000 and 1,600 kg/m^3 where soils with high organic materials and a mix of certain clay minerals can have bulk densities less than 1,000 kg/m^3. Sediment bulk densities are useful in dating and understanding the trends of radionuclides along the sediment cores (Putyrskaya *et al.*, 2015).

4.2.2 Distribution of Artificial and Natural Radionuclides in Sediments

The multiple radionuclides namely; ^{137}Cs, ^{40}K, ^{210}Pb, ^{214}Bi and ^{214}Pb activities were detected in Crescent (CR) and Malewa (M1) Cores within a single measurement in the detector. The activities of these radionuclides along the sediment cores are presented in Figures 4.5 (a) and (b). The soils have a high K concentration and ^{40}K which is a naturally occurring isotope.

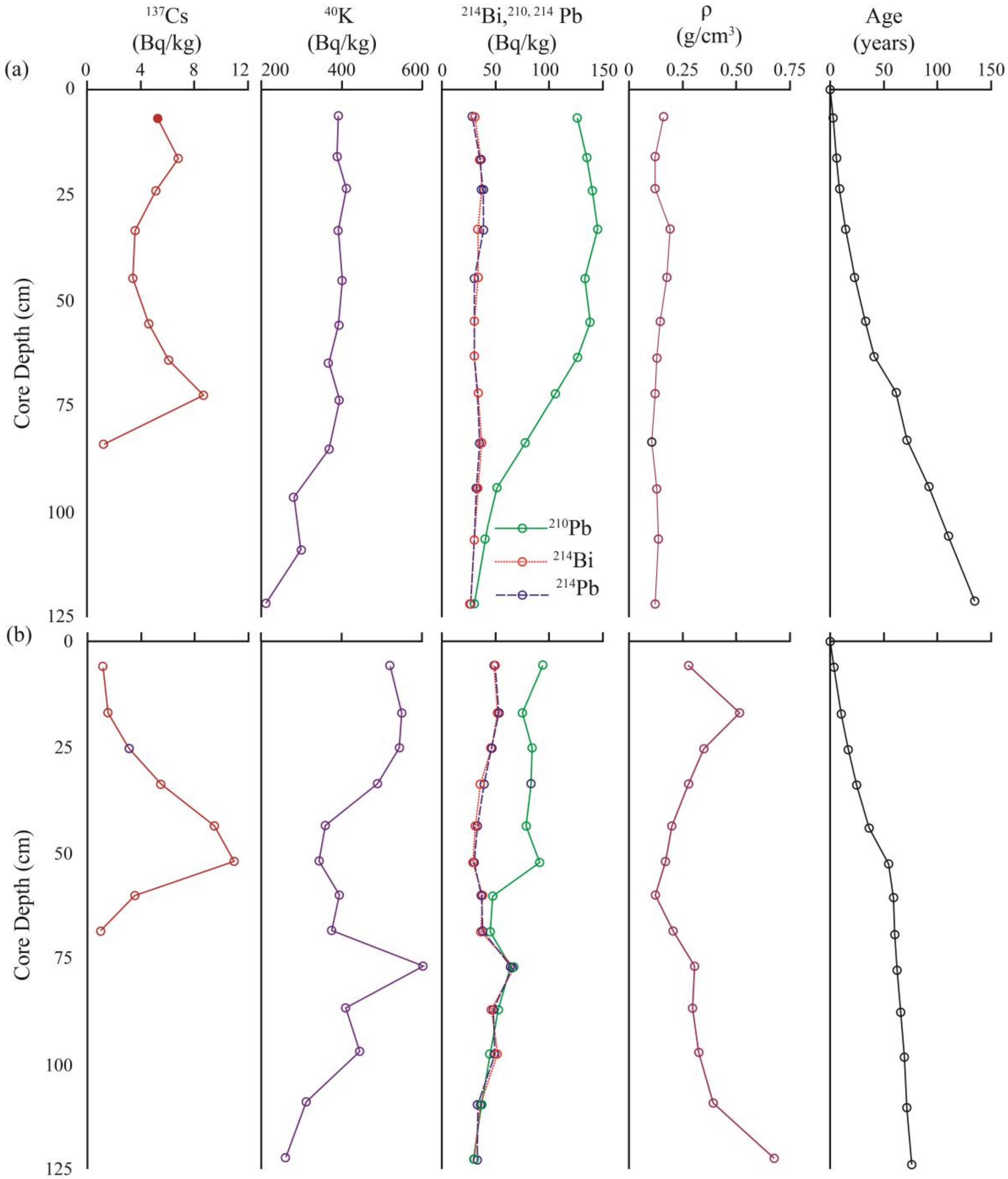


Figure 4.5: Distribution of radionuclides activities, bulk density (ρ) and sediment age along the Lake Naivasha sediment cores (a) Crescent site – CR core and (b) Malewa site – M1 core.

The ^{40}K and ^{137}Cs are alkaline elements with similar chemical properties. As a result, their diffusion behaviour can be assumed to be similar too (Grado, 2015).The ^{137}Cs activities ranged between 1.44 - 8.77 Bq/kg and 1.01 – 10.99 Bq/kg for CR and M1 cores, respectively. The ^{137}Cs concentration was found to be low a finding that agreed with those reported by Matisoff (2014) who reported that there was low ^{137}Cs activities in the southern hemisphere. Low ^{137}Cs activities with a range of 0.00 – 4.28 Bq/kg in both depositional and disturbed sites had also been reported by Jeff and Shepherd (2009) in a study conducted on soils in Saiwa River watershed, Western Kenya. In 2016, during the Lake Naivasha radionuclide studies, it was observed that ^{137}Cs peak activity was detected at 72 cm and 52.5 cm depth for CR and M1 cores, respectively. These peak activities were considered to represent year 1963 which registered the maximum deposit from the nuclear weapon testing (IAEA, 1998; Putyrskaya and Klemt, 2007; Ruecker *et al.*, 2008; Apostu *et al.*, 2012; Mabit *et al.*, 2014d). There was no ^{137}Cs activity that was detected beyond 83 cm and 69 cm for CR and M1 cores, respectively. These depths, 83 cm and 69 cm were considered to mark the emergence of nuclear weapon testing which was in early 1950s.

It was also observed that the ^{214}Bi and ^{214}Pb activities along CR and M1 sediments cores were closely related (Figure 4.5). As a result, to calculate the excess ^{210}Pb in the studied cores, an average of ^{214}Bi and ^{214}Pb activities in each core was used. It was found out that the maximum value of excess ^{210}Pb did not exist at the surface layer of the cores as shown in Figure 4.5. Similar observations had been made by Yang and Turner (2013) and Putyrskaya *et al.* (2015) when dating sediment cores from various lakes of China and Europe, respectively. In addition, an irregular profile of ^{210}Pb for CR and M1 core was observed. A similar trend had been reported by Stoof-Leichsenring *et al.* (2011) for a dated sediment core of Lake Naivasha. Cases where ^{210}Pb activity was maximum in subsurface layers were also reported by Putyrskaya *et al.* (2015) for sediment cores collected in Lakes Brienz, Thun, Lucern and Prealpine Lake Meggione of Switzerland.

Wang *et al.* (2016) reported a similar trend for ^{210}Pb on sediment cores collected from Karnaphuli River estuary, Chittagong, Bangladesh.

According to Appleby and Oldfield (1978), low ^{210}Pb activity recorded at the upper sections of sediment core, can be attributed to mixing of sediments experienced from wind wave action. Post depositional redistribution, can be experienced in cases where sediment mixing takes place (Verschuren, 2001). According to Miguel *et al.* (2003) activities of unsupported ^{210}Pb, are largely affected by rapid mixing experienced at the surface layers of sediment cores. The radionuclides that are in the lakes/reservoirs bed may be redistributed within the sediment column due to physical or biological mixing at the sediment-water interface. It can also be redistributed because of chemical diffusion within the water pores. In cases where sediment mixing has taken place, a flattening of ^{210}Pb profile is experienced in the surficial sediment layers (Appleby and Oldfield, 1978).

After assessing the ^{210}Pb profile, (Figure 4.5) it was found out that the simple ^{210}Pb models such as CF-CS and CIC models could not be used in assigning of dates to the sediment layers. As a result, dates from CRS-Pw model where ^{137}Cs was used as an independent marker, were found to be reliable for the current study. According Appleby and Oldfield (1978) CRS model is applicable in cases where mixing has taken place. This is because use of uncorrected ^{210}Pb dates in a sediment core with a mixing zone spanning close to 10 years accumulation, the maximum error on dates assigned is usually less than 2 years. The ^{137}Cs was successfully used by Miguel *et al.* (2003) and Putyrskaya *et al.* (2015) as an independent marker while applying CRS-Pw model in dating cores collected from different lakes in Switzerland.

4.2.3 Geochronology of Sediment Cores

From ^{137}Cs profile it was possible to assign one date corresponding to year 1963 at the nuclide's peak activity layer (72 cm and 52.5 cm depth for CR and M1 cores, respectively). This is because the study was conducted in the southern hemisphere, which was not affected by Chernobyl accident, in year 1986. This is also used as a reliable marker in northern hemisphere. The surface of the sediment core was assumed to correspond to the date of sampling since ^{7}Be radionuclide was not measured for the current study. The CR and M1 sediment cores of 129.5 cm were assigned dates up to 140 ±10 and 81 ±7 years, respectively (Figure 4.6). These dates were found to be within the acceptable application period of ^{210}Pb geochronological analysis.

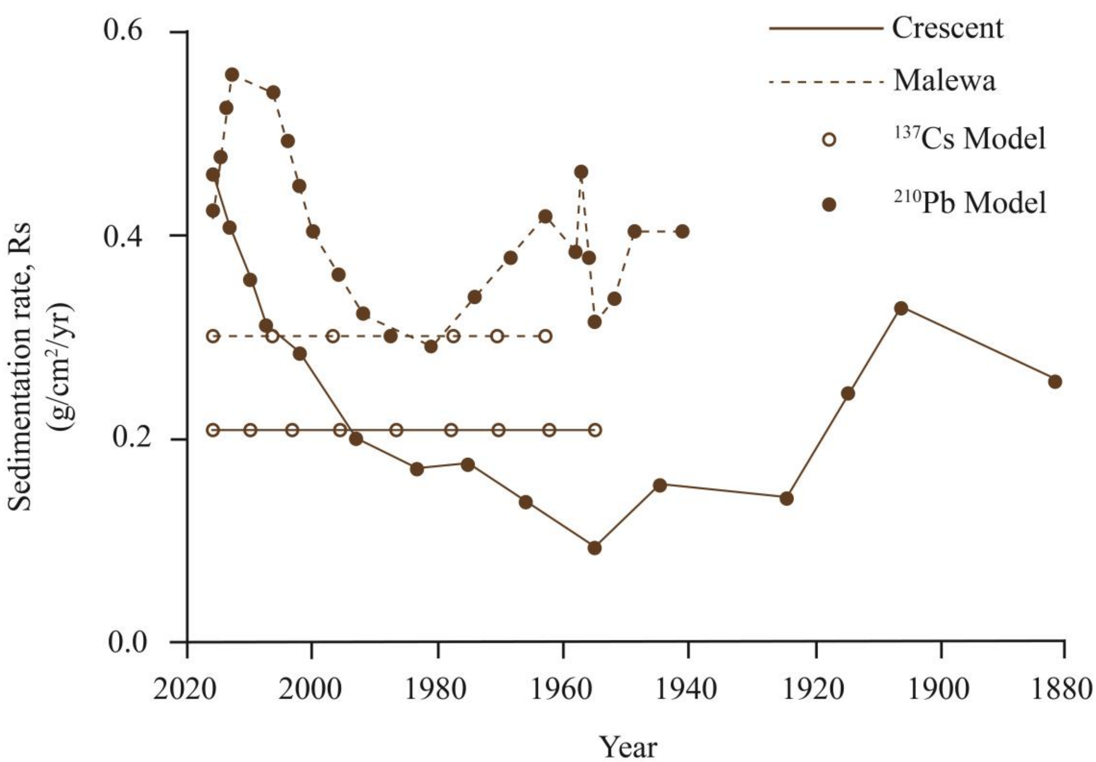


Figure 4.6: Sedimentation rate at Crescent (CR core) and Malewa (M1 core) sites in Lake Naivasha, derived from ^{137}Cs and CRS-Pw ^{210}Pb models

Appleby and Oldfield, (1978); Steiner *et al.* (2000); and Putyrskaya *et al.* (2015) observed that application of ^{210}Pb radionuclide in dating is limited to the past 100-150 years which is approximately 5 to 7 times its half-life (22.6 years). In this study, the use of ^{210}Pb CRS-Pw model aided in assigning of dates and determining the recent mass sedimentation rates of Lake Naivasha. Miguel *et al.* (2003) reported that ^{210}Pb radionuclide nuclide has been used extensively in estimating the sedimentation processes and rates. Hence, from the dates assigned on Lake Naivasha sediment cores, layers corresponding to the past 20 and 50 years were identified and used to establish sediment thickness, volume and distribution from the multifrequency acoustic images.

4.2.4 Sediment Thickness, Distribution and Volume

A variable sediment thickness was observed in various parts of Lake Naivasha as shown by the sediment isopach given in Figure 4.7. Location of the cross-sections are provided in Figure 4.1.

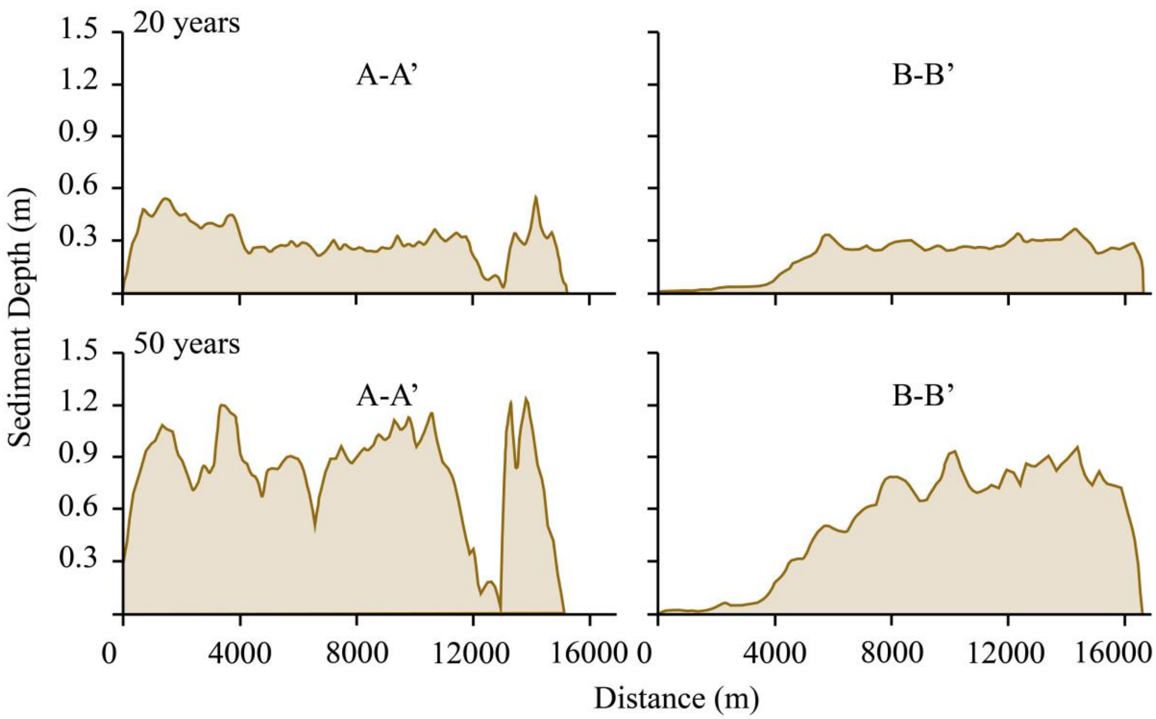


Figure 4.7: Sediment Isopach in the past 20- and 50-years period. A-A' west - east and B-B' north - south direction of Lake Naivasha, Kenya

Sediment thickness ranging from 0.15 m to 0.55 m throughout the lake was recorded for the past 20 years while a range of 0.24 m to 1.9 m thickness above the 20 years layer presented sediment deposited in the past 50 years. Tarras-Wahlberg *et al.* (2002) had reported that sediment thickness of 10 - 20 cm in the mid lake might have been deposited in the past 10 - 15 years. The sediment thickness for the past 20 years was found to be on the increase. The increase in sediment thickness can be attributed to the influence of anthropogenic activities within the basin. A similar finding was reported by Schmengler and Vlek (2015) for a study conducted in Wahble and Fafo reservoirs.

Sediment deposition variation within Lake Naivasha is as presented by sediment isopach profiles (Figure 4.7). It was found out that the mean sediment thickness in Lake Naivasha for the past 20 and 50 years was 0.25 m and 0.56 m, respectively. The mean sediment thickness at different parts of Lake Naivasha are presented in Table 4.2. It was observed that, the thickest sediment deposits in Lake Naivasha are at Crescent Island Lake with a maximum sediment isopach of 0.57 m and 1.88 m for past 20 and 50 years, respectively.

Table 4.2: Mean sediment thickness observed in various parts of Lake Naivasha between 1966 – 1996, 1996 – 2016 and 1966 – 2016

Site	**1966 – 1996** **≈ 30 years**	**(1996 – 2016)** **≈ 20 years**	**(1966 – 2016)** **≈ 50 years**
North	0.22	0.23	0.45
East	0.6	0.32	0.92
South	0.44	0.3	0.74
West	0.48	0.37	0.85
Middle	0.52	0.26	0.78

It was observed that high sediment deposition mainly occurred near Hippo point (Figure 4.8) for the main lake and at Crescent Island Lake. These parts of the lake are characterized by water depth of 6 – 8 m and 12 – 17 m at Hippo Point and Crescent Island Lake, respectively (Appendix 4). Sediment thicknesses of 0.37 m and 0.85 m were recorded at hippo point for the past 20 and 50-years period, respectively. According to Verschuren (1999a) parts of Lake Naivasha with water depth greater than 6 m are less affected by resuspension and redistribution processes and hence sediments tend to accumulate in these parts. Also, the western part of the main lake is slightly sheltered from strong wind wave action (Everard *et al.*, 2002) and this indicates that at this section of the lake could be a preferential sediment deposition zone.

The sediment isopach maps (Figures 4.8, 4.9 and 4.10) show that sediment accumulation zones for the past 20 and 50 years vary. It was observed that in the past 20 years, less sediment was deposited on northern part of Lake compared to the middle, western, southern and Crescent Island Lake (Figure 4.8). This could be because the northern part of the lake, is shallow (1 - 3 m deep) and therefore the sediment is continuously resuspended and redistributed by wave action (Eggermont *et al.*, 2007) resulting to less sediment deposition. According to Ndungu *et al.* (2015) a strong wave action and current is experienced near the northern part of Lake Naivasha. The pattern of sediment deposition into a lake or reservoir is influenced by the hydrodynamics of a lake, direction and strength of wind (Shotbolt *et al.*, 2005; Ndungu *et al.*, 2015). According to Ji and Jin (2014) oscillatory motion is induced by wind waves in shallow water resulting to an increase in bottom shear stress. This results in softening of sediment bed hence encouraging sediment resuspension.

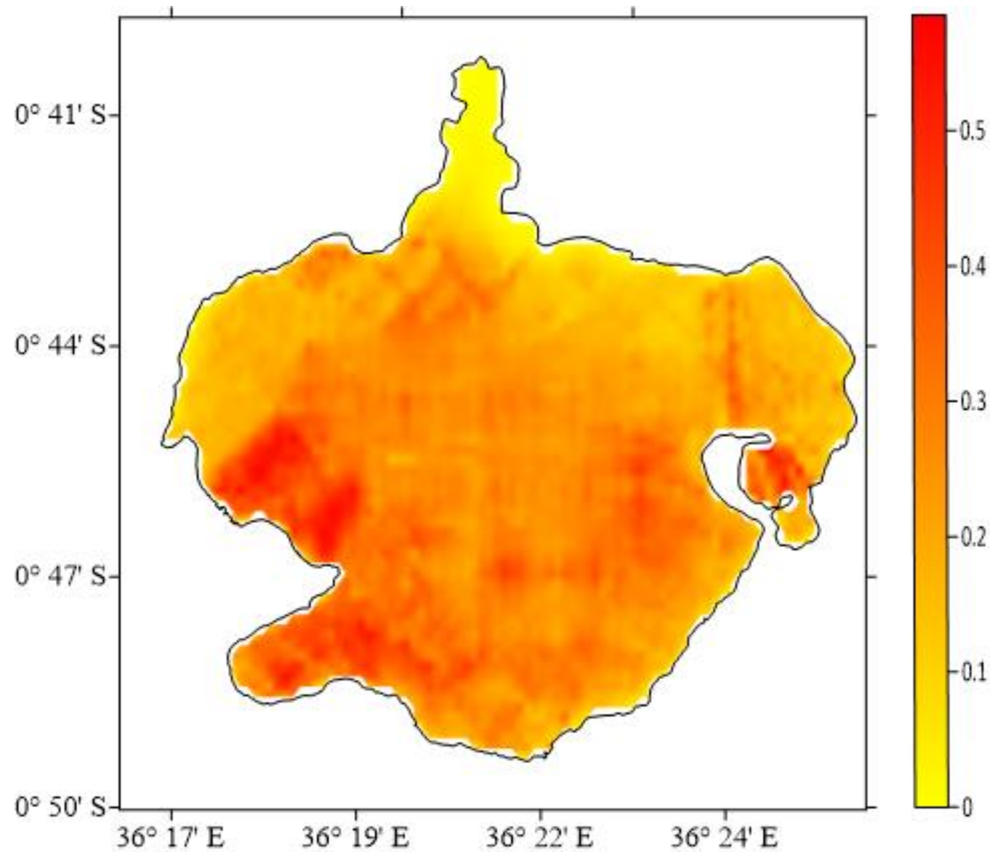


Figure 4.8: Sediment thickness and distribution in Lake Naivasha over the last 20 years (1996 – 2016)

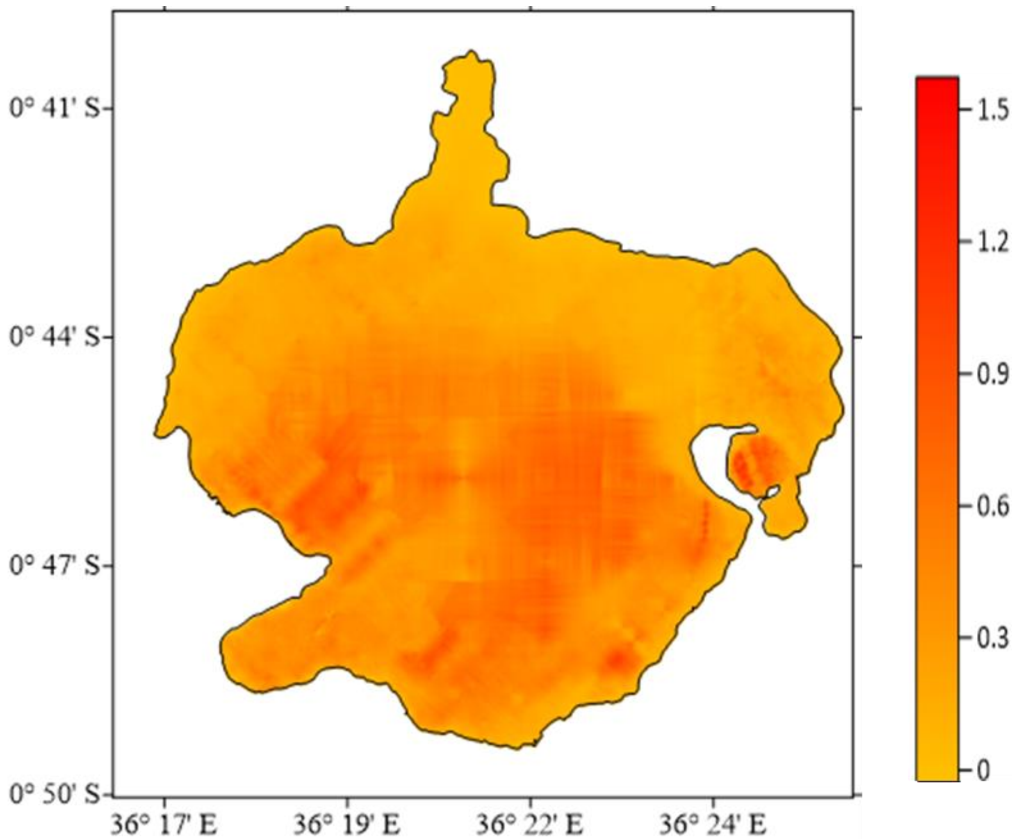


Figure 4.9: Sediment thickness and distribution in Lake Naivasha over the initial 30 years (1966 – 1996)

Though the deposition zone is at the deeper parts of the lake, the actual points of sediment deposition in the last 20 years between 1996 and 2016 (Figure 4.8) and first 30 years between 1966 and 1996 (Figure 4.9) vary. A study conducted by Ndungu *et al.* (2015) reported that the wind wave action in Lake Naivasha varies and this is a phenomenon that can affect the sediment resuspension and redistribution. According to Lehman (1975) sediment deposition patterns in a lake or reservoir varies with time, biological mixing and sediment composition.

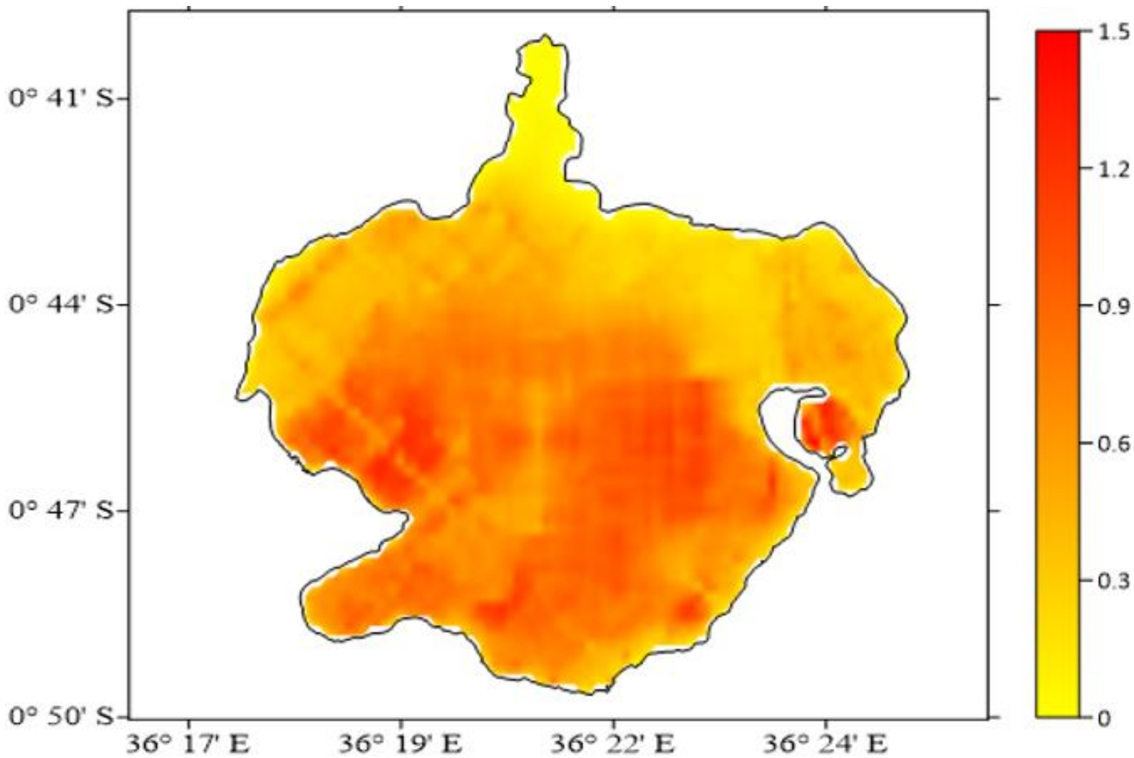


Figure 4.10: Sediment thickness and Distribution in Lake Naivasha for the last 50 years (1966 – 2016)

The findings on sediment thickness and distribution suggest that sediment focusing takes place in Lake Naivasha where the deposited sediment is resuspended and transported from shallower to deeper zones of the lake. Further, it was observed that sediment deposition zones on deeper parts of the lake would be suitable for environmental archives as explained by Shotbolt *et al.* (2005). Sediment collected from these sites would be useful in establishing sediment accumulation rates.

4.2.5 Long-Term Volumetric and Sediment Accumulation Rates from Sediment Coring and multifrequency Acoustic Profiling System Data

The average mass sedimentation rate of Lake Naivasha established from CRS-Pw model for ^{210}Pb was approximately 0.32 g/cm^2/yr. It was found out that the average mass sedimentation rate towards the inflow, represented by M1 core, was 0.40 ± 0.04 g/cm^2/yr. At the Crescent Island Lake, (CR core) a lower mass sedimentation rate of 0.26 ± 0.15 g/cm^2/yr was recorded. A higher sedimentation rate at M1 compared to CR core could be explained by the fact that M1 core was collected near the inflow part of the lake while Crescent is located far from the inflow.

From the sediment core profiles, it was observed that there is a general increase of mass sedimentation rate in recent years. In addition, a large fluctuation of sedimentation rate along M1 core could be attributed to the core being collected close to the inflow part of the lake an area that is easily affected by changes in sediment inflow. Less fluctuation of sedimentation rate was recorded from the CR core. This could be because CR core was collected at the deepest part of Lake Naivasha which is shielded from wind and is far from the inflow. As a result, the sediment in this part of the lake is not easily redistributed. Sediment received at the inflow part of the lake may be distributed to various parts of the lake but the sediment that finds its way to Crescent Island Lake, where CR core was collected, majorly settles there hence the gradual accumulation rate experienced at the site.

A constant sedimentation rate of 0.21 and 0.3 g/cm^2/yr was established while considering the ^{137}Cs radionuclide for CR and M1 core, respectively. This rate was found to be lower than those recorded using ^{210}Pb dating which could be due to the whole of the past 50 years being grouped as one layer. The established sedimentation rate at Crescent Island Lake was found to be comparable to what was reported by Verschuren (2001). Further, from the 2016 and previous study by Stoof-Leichsenring *et al.* (2011), it was observed that the mid lake has the lowest sedimentation rate of 0.08 g/cm^2/yr compared to Malewa and Crescent Island Lake.

From multifrequency Acoustic Profiling system, it was observed that an increase in Lake Naivasha sedimentation has taken place in the period between 1996 - 2016 compared to 1966 – 1996 period. A summary of these findings in presented in Table 4.3.

Table 4.3:Lake Naivasha sedimentation status for periods between 1966-1996, 1996-2016 and 1966 - 2016

Parameter	Period		
	1966 – 1996 (≈ 30 Yrs)	**1996 – 2016 (≈ 20 Yrs)**	**1966 – 2016 (≈ 50 Yrs)**
Sediment Volume (10^6 m^3)	53.4	36.36	89.76
Average dry bulk density (t/m^3)	0.13	0.16	0.15
Volumetric Lake Sedimentation rate (10^6 m^3/yr)	1.78	1.82	1.79
Mean sediment load (10^5 Metric ton/yr)	2.78	4.61	3.96
Sediment accumulation rate (cm/yr)	1.05	1.18	1.12
Sediment yield (10^5 t/yr)	2.38	2.85	2.66
Specific Sediment Yield (t/km^2/yr)	110.7	132.6	123.72

The sediment accumulation rates established from APS were found to be 1.05, 1.18, and 1.12 cm/yr over the period between 1966 – 1996, 1996 – 2016 and 1966 – 2016, respectively. Tarras-Wahlberg *et al.* (2002) reported that the central and southwest parts of the lake experienced a sedimentation rate of about 1.0 cm/yr. This rate was estimated from the stratigraphic thickness records identified from cores collected from central and southwest parts of Lake Naivasha.

It was found out that specific sediment yield of Lake Naivasha basin was 110.7, 132.6 and 123.7 t/km^2/yr, respectively (Table 4.3). An increase in sediment yield was observed in the recent past which could be due to land use changes experienced within the lake's basin. The findings of this study closely compare with those reported by Stoof-Leichsenring *et al.* (2011). It was observed that the calculated specific sediment yield was lower than the average value reported for Africa. The average specific sediment yield reported for African continent, derived from reservoir sedimentation rates, is about 800 t/km^2/yr (Vanmaercke *et al.*, 2014). With the observed changes in sediment yield and accumulation rate over years, there was need to understand the changes in sediment quality over time. Hence, the quality of the sediment for the past 50 years from the time of survey was determined.

From the chronological analysis of Lake Naivasha, it was found out that 72 cm and 52.5 cm of sediment core length would account for the past 50 years at the Crescent and the main lake, respectively. Thus, sediment cores of 53 and 72 cm length considering the main lake and Crescent Island Lake were used to assess the impact of recent environmental change on the lake. This assessment was established from the measured nutrients and heavy metal concentration along the sediment cores collected.

4.3 Nutrients and Heavy Metal in Lake Naivasha

The spatial and temporal variation of nutrients and heavy metal concentration was noted within Lake Naivasha. According to Lee and Cundy (2001) in a dated core, vertical variation in contaminant concentration is useful in assessing changes in pollutants loading over time.

4.3.1 Vertical Trend of Heavy Metals and Nutrients in Lake Naivasha Sediment Cores

It was found out that for the six sediment cores from Lake Naivasha, most heavy metals had their own distribution characteristics (Figure 4.11). The location of collected cores is presented in Appendix 1b. According to Al-Mur *et al.* (2017) vertical assessment of sediment characteristics in a core, is useful in establishing historical sequence of pollution and contamination of a waterbody. It was observed that along the sediment cores collected there was variation of sediment characteristics with time (Figure 4.11). This could be due to changes in sediment input temporally and spatially. Shotbolt *et al.* (2005) reported that in lakes and reservoirs, the amount of sediment deposited varies temporally due to difference in rainfall events and between seasons and in response to land use, vegetation or climatic changes. It also varies spatially due to variation of sediment sources within the catchment.

In assessing the sediment heavy metal concentration profile, it was observed that in most cores, the top layers of the cores had high concentration of heavy metals. According to

Chen *et al.* (2012) the surface of sediments to a few centimetres below, reflects the continuous change in contamination level which reflects on anthropogenic impact to the waterbody.

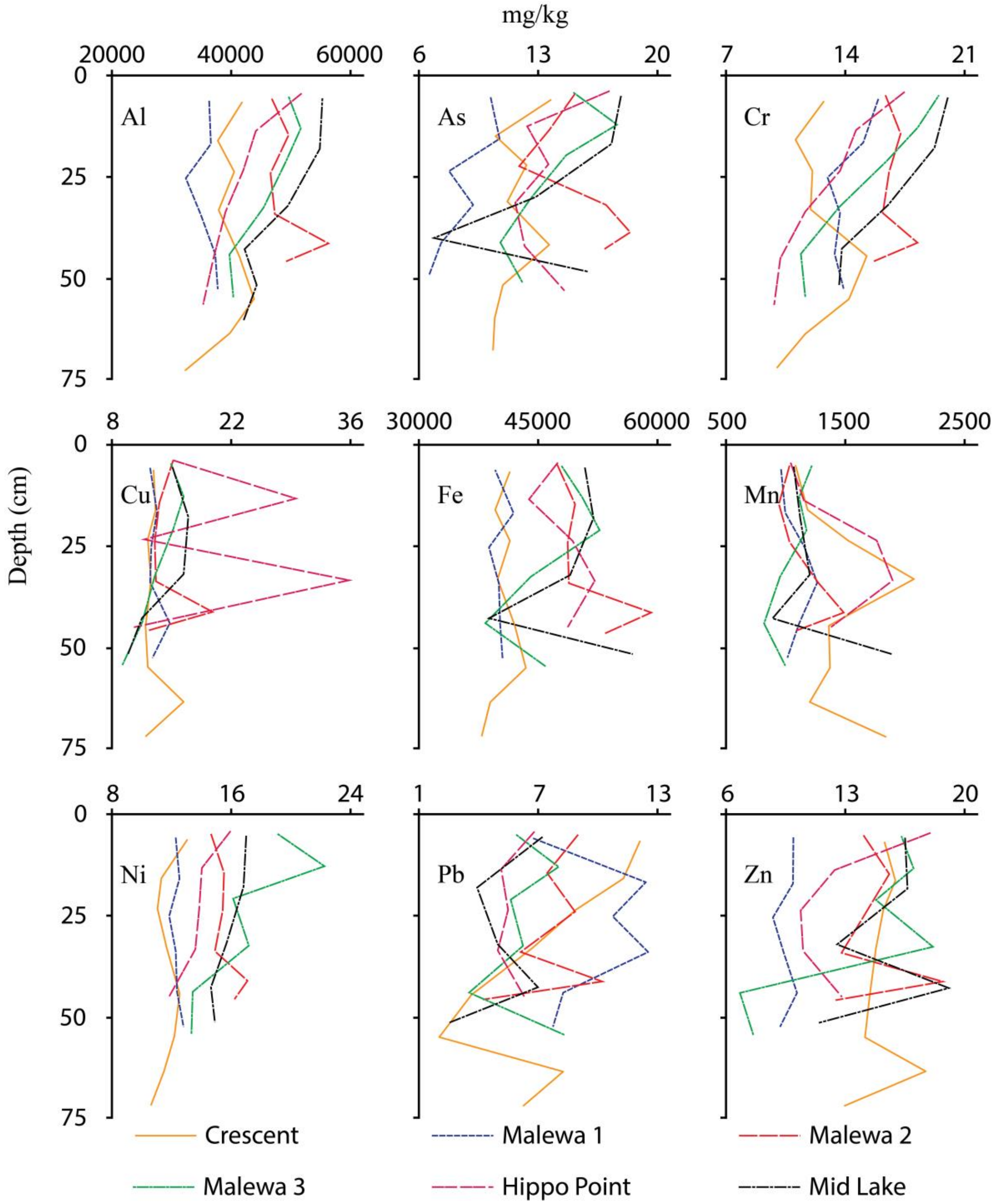


Figure 4.11: Depth wise distribution of heavy metal concentration of six sediment cores collected from Lake Naivasha, Kenya

At the surface of HP and CR core which were collected at Hippo point and Crescent sites (deep parts of the Lake Naivasha) Al and Fe concentration were found to be highest towards the surface of the core. A large fluctuation of Fe affecting the core up to 20 cm depth was observed at Hippo point (HP core). This finding agreed with those of Wang *et al.* (2012) where Fe analysed from top 10 cm of the core sediments, collected from Baihua Lake, fluctuated and had high mean concentration. This is an indicator that an increase in metal delivery into the lake in recent years could be taking place. Al-Mur *et al.* (2017) reported a similar trend on Fe for analysed sediment cores from Red Sea, Saudi Arabia. In Lake Naivasha sediment analysis it was found out that for the rest of cores studied, Al and Fe increased on the second layer and then in deeper layers a general reduction in concentration was recorded. Ni and Fe were found to have almost a similar vertical distribution pattern for most cores studied. According to Zanjani-Jamshidi and Saeedi (2017) when Fe and Ni have similar distribution characteristics, this is usually an indicator that Fe oxides and hydroxides adsorbs Ni.

The Mn was found to greatly fluctuate along all the sediment cores studied especially for layers between 20 and 50 cm. According to Elkady *et al.* (2015) the Mn fluctuation could be due to the diagenetic processes that leads to metal's remobilization. It was also observed that high values of Mn were recorded at Crescent followed by Hippo Point. These parts of Lake Naivasha are less affected by wind action compared with other parts, hence less oxygen is available in sediment which could have led to high amount of reduced Mn. In addition, Mn concentration was found to be higher in the deeper layers of the core than at the surface. In assessing the vertical profiles for Mn and Zn in all the six cores, it was found that the profiles had different patterns which indicated that the Zn availability was not affected by the Mn cycling in the cores. Steiner *et al.* (2000) reported a similar finding where Zn movement was not coupled with that of Mn for a study conducted on an alpine lake in Switzerland. The difference in Zn and Mn movement in the sediments can be attributed to metals being from diverse sources within the catchment (Steiner *et al.*, 2000; Elkady *et al.*, 2015). The Cd was found to be below detection limit in all the sediment cores. Similar findings had been reported by Seshan *et al.* (2010) for sediments in southeast coast of India where low quantity of Cd was detected only in one of the five cores while in the rest it was totally absent.

The vertical distribution of Cr indicates that it is related with that of Fe in M1, M2 and CR sediment cores. Further Cr and Mn were also found to show similar distribution in M2 and M3 sediment cores. Similar observation had been made by Zanjani-Jamshidi and Saeedi (2017) where they had investigated on heavy metal contamination on sediment of Anzali wetland a freshwater ecosystem in the southern part of the Caspian Sea. Same distribution pattern of Cr and Mn indicted the metals are influenced by similar diagenic processes. According to Fu *et al.* (2014) Cr, Co, and Pb fluctuations along the sediment core can be attributed to changes in traffic or agricultural activities. In Lake Naivasha the increase of Cr, Co, and Pb at the surface of sediment cores could be indicating that use of pesticides and fungicides within the catchment is impacting metal concentration in the lake.

In all cores studied, Pb was found to vary with depth but a great variation was observed between the surface and the layer immediately below it. Al-Najjar *et al.* (2011), Yusoff *et al.* (2015) and Al-Mur *et al.* (2017) attributed the vertical fluctuations of heavy metal concentration for instance Pb in sediment cores, to changes of anthropogenic input with time. According to Yusoff *et al.* (2015), variation in Pb concentration could also arise from indirect sources such as atmospheric deposition.

It was observed that from surface to 20 cm depth of sediment cores, As concentration increased in five cores except in M1 core. An increase of As in sediment cores, serves as an indicator of anthropogenic impact to the water body since its major source is from agricultural input (Al-Mur *et al.*, 2017). From the ML core, it was observed that recently mid lake is receiving more heavy metal loading than before. In all metals studied other than that of Mn, the surface of ML sediment core recorded higher metal concentration compared to the lower layers of core. This could be attributed to variation of wind turbulence that the lake experiences (Ndungu *et al.*, 2015) hence transfer of sediment pollutant into middle of lake.

The average phosphorous concentration in Lake Naivasha was found to be 780.5 mg/kg while the minimum and maximum concentration was 448.6 and 1827 mg/kg, respectively. Considering the various sites where the cores were collected, it was observed that the TP concentration spatially and along the sediment cores was varying (Figure 4.12). It was found that high TP concentration was at the Crescent sediment core. Though, sediments that are rich in iron acts as sink to phosphorous, in cases where deoxygenation is experienced within the lakes, phosphorous bound on sediment is released back to the water (Kitaka *et al.*, 2002).

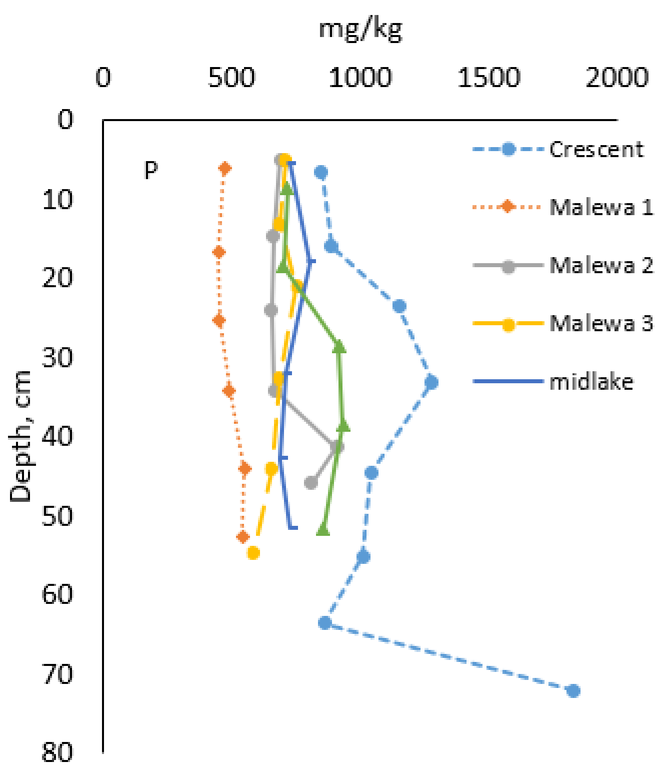


Figure 4.12: Depth wise distribution of phosphorous concentration of six sediment cores collected from Lake Naivasha, Kenya

Further, Crescent Island Lake is located further from the inflow where fine sediment finds its way. Junakova and Balintova (2012) reported that TP concentration is usually higher in reservoir parts where fine sediment particles are deposited. A significant increase of TP was noted to correspond with sediment deposited in 1980s. In Lake Naivasha basin extensive agriculture was adopted since 1980 where even areas on steep slopes were converted into farmlands (Hickley *et al.*, 2002). With intensified agricultural activities and population increase within the basin, extensive application of fertilizer and high generation of domestic sewage is experienced which may be discharged into the lake. According to Zhang *et al.* (2017a) large amounts of TP are discharged into waterbodies by fertilizers and sewage as land use changes and population increase. Also, the total annual TP loads into a lake/reservoir tends to be higher during stormy events.

4.3.2 Concentration and Distribution of Heavy Metals in Lake Naivasha Sediment

The mean concentration of heavy metals in Lake Naivasha sediment for Al, As, Cd, Cr, Cu, Fe, Mn, Ni, Pb and Zn are presented in Table 4.4. The table includes the sediment quality guidelines and the Background values for the various metals which are also known as Global average for shale values. These background values were used in interpretation of the impact of these metals in the sediments. Further, the heavy metal spatial distribution within the lake is presented in Table 4.4.

Table 4.4: Mean concentration and spatial distribution of heavy metals in Lake Naivasha and their background values, together with sediment quality criteria Threshold Effect Level

Variable/ Site	**Al**	**As**	**Cr**	**Cu**	**Fe**	**Mn**	**Ni**	**Pb**	**Zn**
CR	39463	11.68	12.36	12.98	40551	1459	11.73	7.42	172
M1	35868	8.68	14.14	13.16	40144	1086	12.33	9.79	140
M2	52217	16.19	17.72	14.62	54096	1117	16.41	7.99	178
M3	47129	14.18	15.81	14.28	46724	1056	17.59	5.87	169
ML	49260	14.16	16.61	14.02	49544	1233	15.80	5.14	173
HP	42805	13.42	13.52	20.67	48340	1448	13.83	5.77	159
Mean	**44457**	**13.05**	**15.03**	**14.95**	**46566**	**1233**	**14.61**	**7.00**	**165**
Background Values for shale [a]	80000	13	90	45	47200	850	68	20	95
Sediment quality criteria [b]	-	5.9	37	36	-	-	18	35	123

[a] Background values adopted from (Turekian and Wedepohl, 1961)

[b] Sediment Quality Criteria for protection of aquatic life. Threshold Effect Level adopted from (Tarras-Wahlberg *et al.*, 2002)

In all sites studied, Cd in the sediment samples was found to be below detectable limits which indicated low sources of Cd within the basin. Similar findings were reported by Elkady *et al.* (2015) where Cd was found to be below minimum detection limit in all the sites studied. This finding contradicted what was reported by Tarras-Wahlberg *et al.* (2002) in a study conducted in Lake Naivasha and its catchment. Tarras-Wahlberg *et al.* (2002) had reported that the lake sediment contained 6-7 mg/kg Cd values that were significantly higher than global average levels for shale. In their study, they reported that the Cd values recorded could have resulted from analytical errors. On the other hand, it was observed that M2 core recorded highest mean concentration of Al, As, Cr, Fe and Zn. Since the core was collected about 3 km from the inlet of River Malewa, the high metals concentration could be attributed to compounding effect of combined inflow from River Malewa, Gilgil, Karati and the effluent of domestic waste water treatment site (Njogu *et al.*, 2011 & Kitetu, 2011). The CR core collected at Crescent Island Lake was found to have lowest mean concentration of Cr, Cu and Ni. Njogu *et al.* (2011) in an earlier study investigating the distribution of heavy metals in soils, sediment and fish from Lake Naivasha basin had reported that samples collected at Crescent site had low heavy metal concentration.

On the other hand, high mean concentrations of various heavy metals were observed in ML core representing middle part of Lake Naivasha. Yuan *et al.* (2011) had reported similar findings, where middle sections of Taihu Lake in China registered high heavy metal contamination than inflow parts of lake. This could be attributed to impacts of wind wave action on the lake resulting to remixing and redistribution of sediment (Verschuren, 1999b). It was found that the mean concentration of Zn in Lake Naivasha sediment varied from 140-178 mg/kg for all the six cores studied where highest mean concentration was recorded in M2 core closely followed by ML core. The Zn levels in all sites were found to be already elevated since according to Tarras-Wahlberg *et al.* (2002) the threshold effect level of sediment quality criteria is 123 mg/kg. It was also observed that Zn concentration in the sediments were all higher than the background value (95 mg/kg) as presented by Turekian and Wedepohl (1961). The source of Zn is mainly from direct input of municipal effluents, and geochemical weathering of parent rock material (Zahra *et al.*, 2014). According to Tarras-Wahlberg *et al.* (2002) Zn concentration within Lake Naivasha basin is 88 – 190 mg/kg. The study also found that a relatively high Zn concentration in Lake Naivasha catchment could be attributed to natural source. The sources of Zn in reservoirs and Lake sediments can be from batteries, waste water, brass and bronze alloys, fungicides and pesticides (Nzeve *et al.*, 2014). Further, according to Zeng and Wu (2013) Zn, Pb and Cu in aquatic ecosystems can be from industrial wastewaters. However, this could not be the case in this study since Lake Naivasha is not largely surrounded by industries (Tarras-Wahlberg *et al.*, 2002).

The mean As concentration for the various cores studied was found to range from 8.7 – 14.4 mg/kg. As a result, the sediment in Lake Naivasha can be considered to be lowly contaminated with As since the background value for arsenic as reported by Turekian and Wedepohl (1961) is 13 mg/kg. However, the lake would be considered contaminated with As when comparing them with the recommended values in sediment quality criteria; Threshold Effect Level (5.9 mg/kg) in Table 4.4. The main natural and anthropogenic source of As is atmospheric emission and agricultural activities, respectively (Shrivastava *et al.*, 2015). According to Harikumar *et al.* (2009) anthropogenic source of As is from pesticides and herbicides.

It was observed that Al had lowest and highest mean concentration in CR and M2 cores, respectively. Al is an abundant conservative element that is not easily affected by anthropogenic activities (Yang *et al.*, 2014) as a result, the source of Al in Lake Naivasha was attributed to geogenic sources. The Iron (Fe) concentration in most parts of Lake Naivasha was also considered to be largely controlled by geogenic sources. However, the highest Fe concentration was recorded in M2 core with a mean concentration of 54096 mg/kg while the lowest Fe was measured in CR core. The source of Fe in M2 core could be from agricultural activities or the domestic wastewater effluent.

On the contrary, high Mn concentration was recorded on CR core which was collected at the Crescent Island Lake followed by HP core. The high Mn concentration recorded for HP and CR core could be attributed to Mn remobilization into deeper layers. Elkady *et al.* (2015) made a similar observation, of Mn being high in sediments collected from deeper parts of Lake Manzala, Egypt. According to Zeng and Wu (2013) Fe and Mn are more sensitive to redox-reactions than the other elements. Further, under oxidizing conditions, reduced Mn remobilization takes place and it is redeposited near the surface

(Elkady *et al.*, 2015). The geogenic sources of Mn are considered to have greater impact on soils than the anthropogenic sources (Yang and Sañudo-Wilhelmy, 1998). It was found that heavy metal concentration and distribution in Lake Naivasha sediment can be attributed to geogenic and anthropogenic sources.

4.3.3 Sediment Source Identification

From PCA results, it was found that the first three components, with eigenvalue greater than 1 accounted for about 97% of the total variance in the data. The PC1 accounted for about 67.1% and was dominated by Al, As, Cr, Fe and Ni. Further, Mn and Pb were dominant in PC2 that explained about 19.0% variance while, Zn and Cu were represented in PC3 with 11% variance (Table 4.5).

Table 4.5: Variance, eigenvalues and eigenvectors on Principal Component Analysis matrix of heavy metals in Lake Naivasha sediment

	Principal Components (PC)					
Parameter	1	2	3	4	5	6
Al	**0.401**	0.105	0.082	0.051	0.090	0.233
As	**0.544**	0.143	0.316	**0.612**	0.382	0.209
Cr	**0.389**	0.195	0.083	0.058	0.355	0.389
Cu	0.235	0.218	**0.579**	0.377	0.576	0.146
Fe	**0.395**	0.070	0.177	0.237	0.298	0.361
Mn	0.029	**0.715**	0.276	0.569	0.275	0.099
Ni	**0.387**	0.189	0.119	0.306	0.234	0.387
Pb	0.258	**0.704**	0.190	0.034	0.411	0.452
Zn	0.159	0.080	**0.398**	0.050	0.016	0.486
Eigen Values	6.037	1.708	1.041	0.108	0.084	0.022
Proportion (%)	0.671	0.190	0.116	0.012	0.009	0.002
Cumulative proportion (%)	67.077	86.057	97.619	98.816	99.751	1.000

Metals clustered in one component, indicates that they could have similar source and distribution patterns (Fu *et al.*, 2014). Since heavy metals in PC1 were highly associated with Al and Fe, the metals in this group are largely associated to geogenic or geogenic sources. However, heavy metals in PC2 and PC3 were from both natural and anthropogenic sources. The concentration of heavy metals in PC2 are easily affected by the diagenic processes involved. A study conducted by Chen *et al.* (2012) on river sediments reported almost similar metal classification in three different PCs that explained the mixed sources of the metals. In this study, the PCA and Correlation results were found to agree on metals that were placed in each category. The PCA and Correlation analysis were also reported by Al-Mur *et al.* (2017) and Yuan *et al.* (2011) to be successful in distinguishing sources of metals in sediments of Red sea, Saudi Arabia and Taihu Lake in china, respectively.

In this study, strong positive correlation among elements in the sediment were considered to suggest common source or geochemical activities. On the other hand, elements that had strong negative correlation were attributed to having various sources and geochemical activities. A similar classification of correlation results had been reported by Bhuiyan *et al.* (2015); Ghrefat *et al.* (2011) and Wang *et al.* (2012) where strong positive correlation between elements indicated that the elements are from the same source and that their transportation process is similar.

Using Pearson's correlation analysis on nutrients and heavy metals in Lake Naivasha sediment, a strong negative correlation of 0.8 was found between Al, Ni, Mg and Zn metals and phosphorous in sediment. At Crescent, P had a strong negative correlation with Al (-0.772), K (-0.585), Mg (-0.530), Ni (-0.613) and Zn (-0.732) respectively. On the other hand, P was found to have a strong positive correlation with Mn (+0.84). Near the inflow part into the lake at Malewa site it was found that Al (+0.543), Zn (+0.314), and Mn (+0.543) had positive correlation with P while As (-0.711), and Pb (-0.514) had negative correlation. Also, from M2 core, P was found to have a strong positive correlation (+0.632 to +0.864) with all metals except Mn and Pb that had a moderate positive correlation (+0.318 and +0.380). The M2 core had been collected towards the inflow of municipal wastewater. Thus, this is an indicator that the sediments quality have been impacted by anthropogenic activities. At Hippo Point, P was found to have a strong positive correlation with Fe (+0.998), Mn (+0.9) and Mg (+0.6). The Al (-0.6), Cr (-0.6), Ni (-0.6), Zn (-0.6) and Pb (-0.4) were found to be negatively correlated with P (Table 4.6). It was found that the presence of P could be attributed to impact by anthropogenic activities. This is also signified by negative correlation with Al, which had been classified to be from geogenic sources. In general, high nutrient values did not match with those of heavy metals which was indicated by the distinct lack of correlations between them. A similar finding had been reported by Yuan *et al.* (2011) for multivariate analysis that had been conducted on sediment and water collected from Taihu Lake. The lack of correlation could be attributed to the fact that nutrients are highly adsorbed in fine sediment compared to the heavy metals' concentration.

Table 4.6: Correlation coefficient for heavy metals in six sediment cores from Lake Naivasha, Kenya

	Al	**As**	**Cr**	**Cu**	**Fe**	**Mn**	**Ni**	**Pb**	**Zn**	**P**
CR										
Al	1.000									
As	**0.525**									
Cr	**0.821**	**0.636**								
Cu	0.053	-0.360	-0.254							
Fe	**0.888**	**0.562**	**0.867**	-0.397						
Mn	**-0.564**	-0.306	-0.303	-0.392	-0.327					
Ni	**0.739**	**0.730**	**0.750**	-0.085	**0.661**	-0.478				
Pb	-0.186	0.110	**-0.532**	0.317	-0.395	-0.353	-0.024			
Zn	0.286	-0.095	-0.083	**0.882**	-0.158	**-0.548**	0.070	0.498	1.00	
P	**-0.772**	-0.378	-0.477	-0.455	-0.474	**0.84**	**-0.613**	-0.300	**-0.73**	1.0
M1										
Al	1.000									
As	-0.067									
Cr	0.402	**0.751**								
Cu	0.450	-0.265	-0.256							
Fe	**0.632**	0.369	0.343	0.313						
Mn	**-0.579**	-0.234	**-0.724**	-0.028	-0.343					
Ni	**0.876**	-0.168	0.236	0.156	**0.686**	-0.423				
Pb	**0.687**	0.406	**0.549**	**0.623**	**0.514**	-0.414	0.312			
Zn	-0.477	0.379	-0.259	-0.157	0.327	**0.567**	-0.230	-0.267	1.00	
P	**0.543**	**-0.711**	-0.257	0.200	0.086	**0.543**	-0.314	0.314	0.31	1.0
M2										
Al	1.000									
As	**0.780**									
Cr	**0.970**	**0.654**								
Cu	0.450	0.472	0.360							
Fe	**0.987**	**0.814**	**0.926**	0.455						
Mn	-0.034	0.356	-0.182	**0.650**	0.040					
Ni	**0.984**	**0.735**	**0.954**	0.333	**0.986**	-0.118				
Pb	**0.526**	0.159	**0.613**	**0.695**	0.457	0.081	0.451			
Zn	**0.968**	**0.646**	**0.980**	**0.521**	**0.927**	-0.069	**0.935**	**0.707**	1.00	
P	**0.793**	**0.837**	**0.632**	**0.654**	**0.864**	0.380	**0.780**	0.318	**0.69**	1.0
M3										
Al	1.000									
As	**0.954**									

	Al	**As**	**Cr**	**Cu**	**Fe**	**Mn**	**Ni**	**Pb**	**Zn**	**P**
Cr	**0.927**	**0.891**								
Cu	**0.979**	**0.980**	**0.897**							
Fe	**0.930**	**0.844**	**0.800**	**0.933**						
Mn	**0.911**	**0.805**	**0.947**	**0.875**	**0.895**					
Ni	**0.857**	**0.920**	**0.809**	**0.841**	**0.629**	**0.633**				
Pb	**0.848**	**0.872**	**0.682**	**0.823**	**0.686**	**0.563**	**0.944**			
Zn	**0.788**	**0.652**	**0.650**	**0.666**	**0.664**	**0.630**	**0.744**	**0.844**	1.00	
P	**0.639**	0.437	**0.551**	**0.605**	**0.841**	**0.787**	0.153	0.243	0.45	1.0
ML										
Al	1.000									
As	**0.762**									
Cr	**0.987**	**0.675**								
Cu	**0.831**	0.382	**0.825**							
Fe	0.382	0.873	0.247	0.053						
Mn	-0.274	0.396	-0.404	**-0.504**	**0.784**					
Ni	**0.995**	**0.755**	**0.991**	**0.793**	0.355	-0.300				
Pb	0.129	-0.382	0.281	0.176	**-0.728**	**-0.829**	0.167			
Zn	0.092	-0.390	0.231	0.122	**-0.738**	**-0.840**	0.165	**0.726**	1.00	
P	**0.500**	**0.700**	0.200	0.254	**0.900**	**0.500**	**0.500**	**-0.600**	-0.30	1.0
HP										
Al	1.000									
As	**0.901**									
Fe	-0.433	-0.192								
Cu	-0.106	-0.453	0.066							
Cr	**0.985**	**0.844**	-0.478	-0.062						
Mn	**-0.652**	**-0.559**	**0.811**	0.246	**-0.598**					
Ni	**0.948**	**0.819**	-0.219	0.090	**0.952**	-0.398				
Pb	0.482	**0.752**	-0.135	**-0.740**	0.356	-0.619	0.268			
Zn	**0.801**	**0.855**	-0.357	-0.323	**0.692**	**-0.816**	**0.633**	**0.841**	1.00	
P	**-0.600**	**-0.500**	**-0.60**	0.100	**0.998**	**0.900**	**-0.60**	-0.40	**-0.60**	1.0

Considering the heavy metals, Al and Fe were found to be strongly correlated (0.6 to 0.987) in sediment from various cores collected. This was an indicator that the presence of these elements in Lake Naivasha sediment was mainly from geogenic source. A similar observation of Al and Fe being from geogenic source had reported for sediment cores collected in Malaysia by Yusoff *et al.* (2015). It was observed that, a strong negative correlation of existed on Mn and Al (Table 4.6) in five (CR, M1, HP, ML and M2 core) of the cores studied. This finding implied that Mn concentration have been influenced by anthropogenic input or was affected by diagenetic processes. According to Elkady *et al.* (2015) strong negative correlation between Mn and Al could be because of geochemical processes acting on sedimentary Mn.

It was found that at Crescent site Al, As, Cr, Fe and Ni had a strong positive relationship a fact that supported the PC1 group results. A strong positive correlation of Pb with Cu and Zn was observed while Zn and Mn had a strong negative correlation (Table 4.6). These findings closely agreed with the classification of PCA results. It was observed that a negative relationship existed between Al and Fe at Hippo point (HP core). This indicated that Fe in Hippo point could be from both natural and anthropogenic source. At Hippo point, it was also observed that Fe and Mn had a strong relationship (0.8) a region where Mn had been found to be of high quantity. On the other hand, a negative relationship of Fe and Mn in Crescent (CR core) and the inflow part of the lake (M1, M2 and M3 cores) was observed. This indicated that the two metals (Fe and Mn) could be from various sources or the processes affecting their distribution are not similar. The findings of this study were different from those reported by Bhuiyan *et al.* (2015) who observed that Fe and Mn had moderate positive relationship of 0.494 in sediments while in water almost no correlation existed between the two metals.

In M1 core, a strong negative relationship between Al and Mn was noted. Since Al was classified to be from geogenic sources, the Mn and Pb (-0.41) were considered to have been influenced by anthropogenic activities. The Zn and Mn were found to have a strong negative relationship (-0.816) at hippo point (represented by HP core). Also, a negative relationship existed between Zn and Fe together with Cu. This was an indicator that these metals could have been from diverse sources and that they may experience different diagenic processes (Elkady *et al.*, 2015). Cu and Zn were found to have strong positive correlation of 0.882 and 0.623 in CR and M1 core, respectively. This suggested that these elements in both sites could influenced by anthropogenic activites. Further, in HP, M3, ML and M2 core, Zn and Pb were found to have a strong positive correlation (>0.7) which could be an indicator that the factors affecting distribution of these metals are the same. According to Tarras-Wahlberg *et al.* (2002); Ndungu *et al.* (2014) wastewater or agricultural sources especially fungicides and other agrochemicals used in the farms surrounding the lake could be the source of Cu and Pb in Lake Naivasha. This classification of Cu and Pb to anthropogenic sources agreed with a finding that they had no correlation with Al which was classified to be from geogenic sources.

Bhuiyan *et al.* (2015) and Ghrefat *et al.* (2011) reported that metals which had no significant correlation with Al, would be suggesting that they are from anthropogenic sources. Since Al in Lake Naivasha was found to be majorly from geogenic sources, it was thus considered to be the most suitable metal for normalization procedure. Normalization of metals is useful in understanding the level of contamination in the heavy metal under consideration. According to El-Amier *et al.* (2017) the normalizing metal should not be easily affected by anthropogenic inputs.

4.3.4 Contamination of Lake Sediments

The chronological variation of contamination level for Crescent and Malewa sites are presented in Figures 4.13 and 4.14. High values of contamination indices were observed at the top or just at the subsurface of the core. This indicated that in the recent past metal contamination has been on the increase. From contamination indices (CF, EF and I_{geo}) it was found that As, Cr, Ni and Zn metals contamination had increased slightly from the past to recent years (Figures 4.13 and 4.14). Yusoff *et al.* (2015) had reported similar findings where EF values of Zn, Cu and Ni had increased from the past to recent years for a study conducted in Tanjung Pelepas harbour.

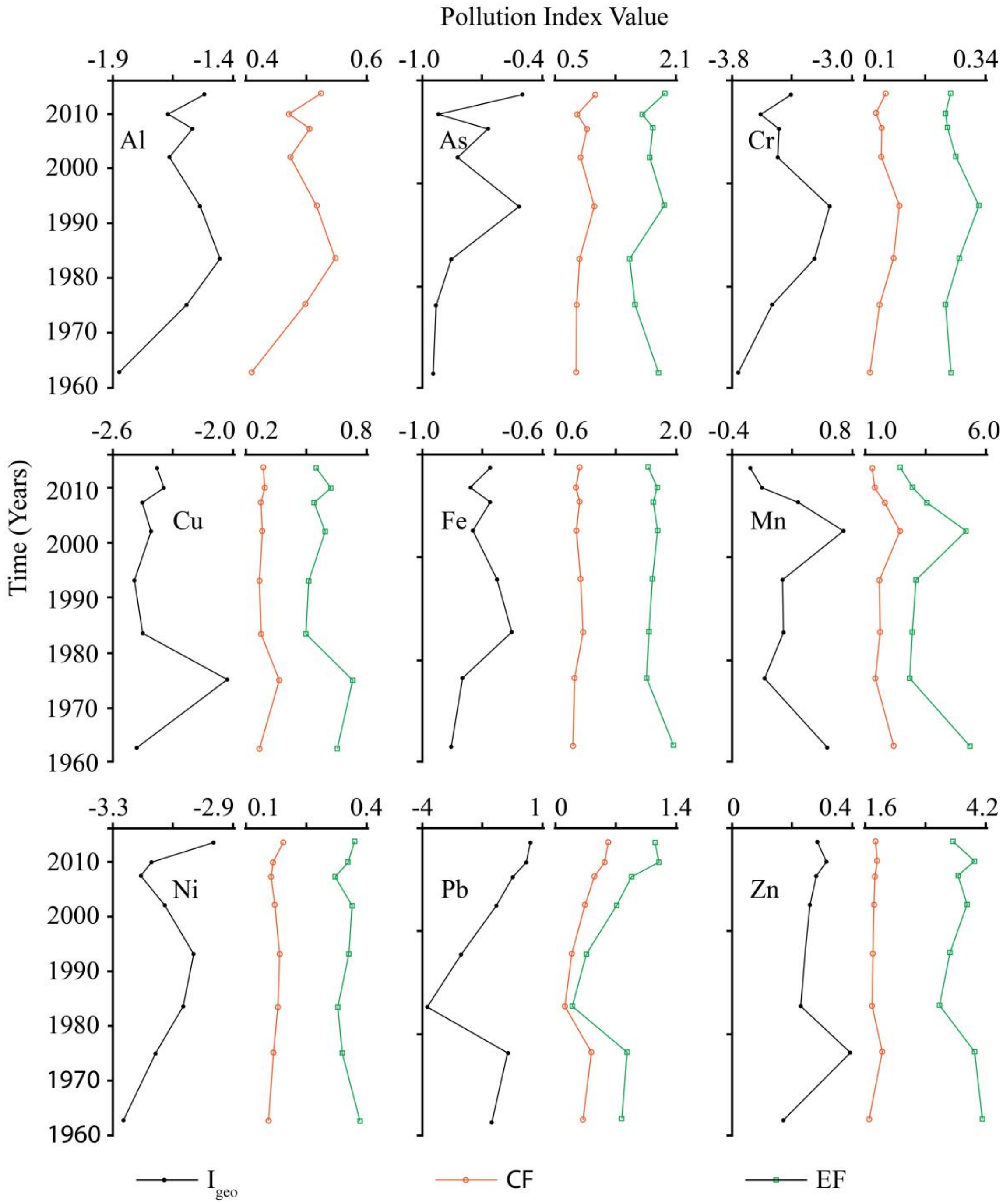


Figure 4.13: Pollution levels for various heavy metals in sediment at Crescent site of Lake Naivasha derived from pollution indices

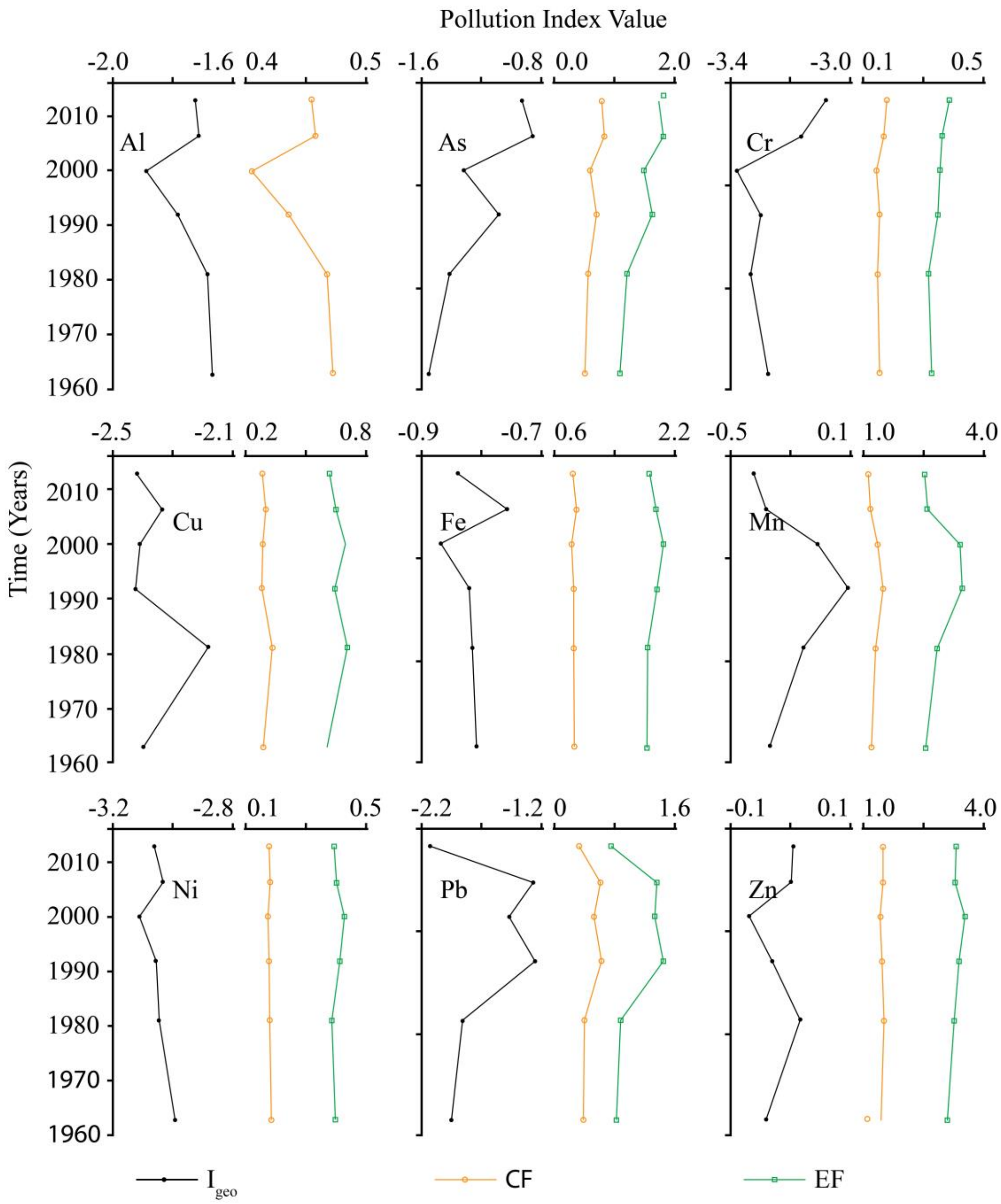


Figure 4.14: Pollution levels for various heavy metals in sediment at Malewa site of Lake Naivasha derived from pollution indices

Further, at Hippo Point it was found out that the sediment samples were highly contaminated with Fe and Cu while at Crescent Island Lake contamination was by Mn and Zn elements (Figure 4.15). Considering the I_{geo}, it was found out that the values vary from one metal to another in the different sampling sites of the lake. It was observed that most metals in the Lake had I_{geo} values being less than zero other than the values for Zn in all sampling sites and Mn in hippo point and Crescent. This indicates that Mn and Zn elements in these sites have been influenced by anthropogenic activities.

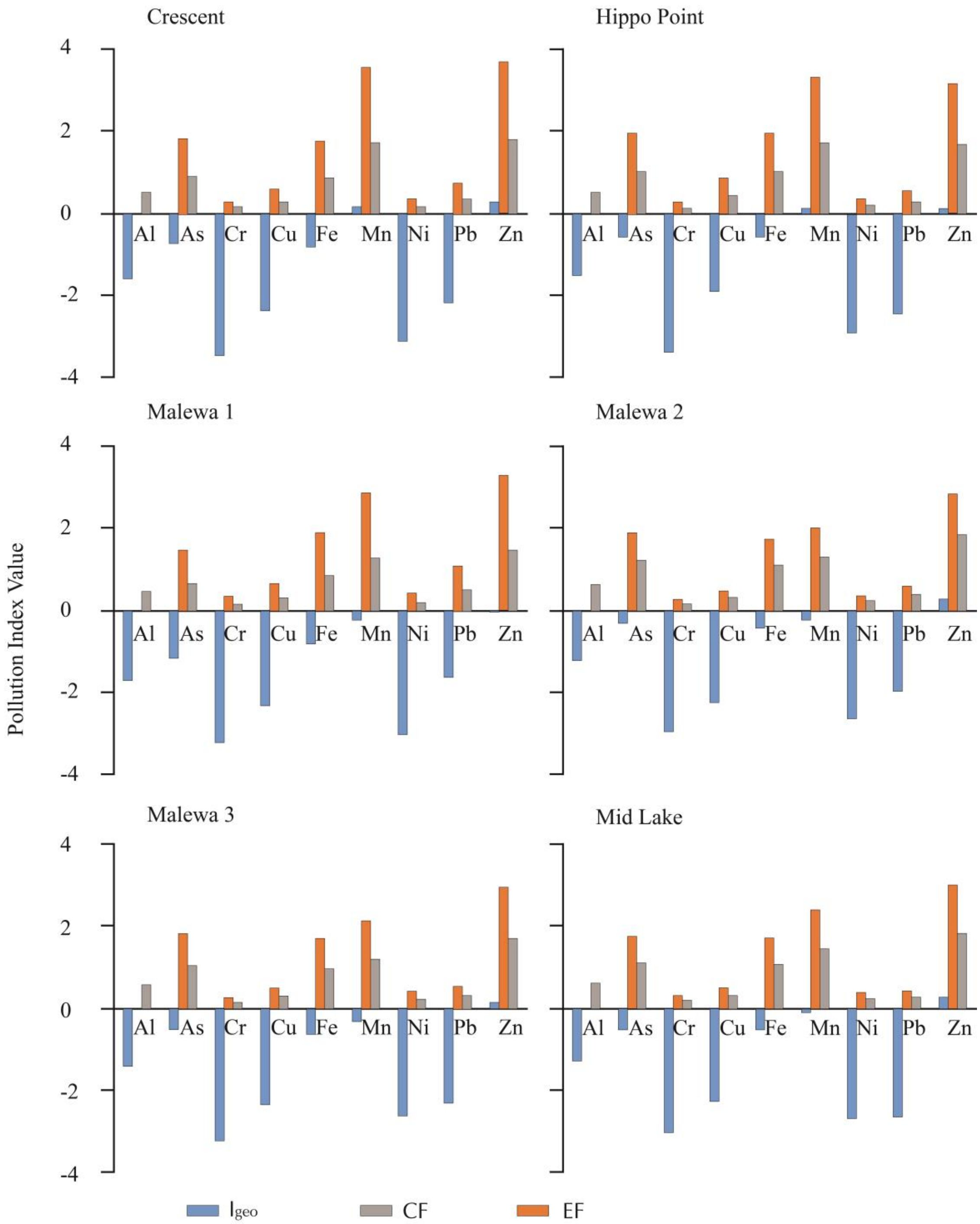


Figure 4.15: Spatial distribution of Contaminant Factor (CF), Enrichment Factor (EF) and geoaccumulation Index (I_{geo}) values for various heavy metals in Lake Naivasha sediment

The Zn I_{geo} values ranged from 0.0 to 0.3 with the high values being reported in M2, ML, and CR cores while the lowest value was registered by M1 core. When considering I_{geo} pollution index, Lake Naivasha was classified in class zero which signify that the sediments are uncontaminated (Table 4.7). This finding indicates that the elements in Lake Naivasha are mainly from natural sources and that the lake was only polluted with Zn and Mn metals.

Further, the trend of Lake Naivasha metal contamination using I_{geo} was similar to that of CFs index. The Al, Cr, Cu, Ni and Pb metals in all cores had CF values less than one (< 1) signifying low contamination. Goher *et al.* (2014) had also reported low CF values for most metals in surface sediments of Lake Nasser Egypt. The As in M2, M3, ML and HP, Fe in M2, M3 and HP core, together with Zn and Mn for all sites were found to have CF values greater than one (Table 4.7). This indicated that the heavy metal concentration in the sediments is not only dependent on geogenic sources but were affected by anthropogenic activities.

On the other hand, Enrichment Factors (EFs) of the sediments were also established to further understand Lake Naivasha heavy metals pollution. In decreasing order, the mean EFs for sediments in Lake Naivasha was Zn >Mn >Fe (As) >Pb >Cu >Ni >Cr with values of 3.2, 2.7, 1.8, 0.68, 0.6, 0.4 and 0.3, respectively (Table 4.7). It was observed that Lake Naivasha is mainly polluted with Zn, Mn Fe and As where the range of Enrichment Factors (EFs) for As (1.5 – 1.9), Fe (1.7 – 1.9), Mn (2.0 – 3.5) and Zn (2.9 – 3.7) in all sites were greater than 1.5 (Table 4.7). Metals with EF values > 1.5, are considered to have been impacted by anthropogenic input (Gao and Chen, 2012; Goher *et al.*, 2014). The EF values for metals along the sediment core were found to be varying. This was an indicator of change in input metal concentration to the sediment depending on their source (El-Sayed *et al.*, 2015). The variation in metals along the sediment core can also be influenced by different diagenic processes that were involved for each metal (El-Sayed *et al.*, 2015; El-Amier *et al.*, 2017). The highest pollution of As, Fe, Mn and Zn was found in M2, both M1 and HP (for Fe), CR and CR cores respectively. It was found out that Enrichment Factor of Zn in all sites is ≥ 3 and this was classified as having moderate enrichment (Table 4.7). Kouidri *et al.* (2016) had also reported moderate enrichment of Zn for sediment from Ain Tomouchent, Algeria.

Table 4.7: Pollution indices classes (adopted from El-Amier et al., 2017; El-Sayed et al., 2015; Zahra et al., 2014) and contamination categories of heavy metal in Lake Naivasha sediment based on Contamination Factor (CF), Enrichment Factor (EF) and Geoaccumulation Index (Igeo)

Indices	Index Value	Class/Level	Sediment description	2016 study (avg)
CF	<1		Low Contamination	Al (0.55), Cr (0.17), Cu (0.33), Ni (0.21), Pb (0.35)
	1 ≤ CF < 3		Moderate contamination	As (1.00), Fe (1.01), Mn (1.45), Zn (1.73)
	3 ≤ CF < 6		Considerable contamination	
	CF ≥ 6		Very high contamination	
EF		*Level*		
	< 1	I	No enrichment	Cr (0.30), Cu (0.61), Ni (0.39), Pb (0.66)
	1 - 3	II	Minor enrichment	As (1.79), Fe (1.80), Mn (2.71)
	3 - 5	III	Moderate enrichment	Zn (3.23)
	5 - 10	IV	Moderately severe enrichment	
	10 - 25	V	Severe enrichment	
	25 - 50	VI	Very severe enrichment	
	>50	VII	Extremely severe enrichment	
I_{geo}		*Class*		
	< 0	0	Uncontaminated	Al (-1.46), As (-0.64), Cr (-3.21), Cu (-2.25), Fe (-0.62), Mn (-0.09), Ni (-2.84), Pb (-2.20)
	0 < 1	1	Uncontaminated to moderately contaminated	Zn (0.19)

Indices	Index Value	Class/Level	Sediment description	2016 study (avg)
	1< 2	2	Moderately contaminated	
	2 < 3	3	Moderately to strongly contaminated	
	3 < 4	4	Strongly contaminated	
	4 < 5	5	Strongly to Extremely contaminated	
	≥ 5	6	Extremely contaminated	

Considering the EF results, it was observed that metals in Lake Naivasha sediment could be from either anthropogenic or geogenic sources. It was found out that Zn was affected by anthropogenic activities such as disposal of municipal wastewater and agricultural inputs like fertilizers and pesticides. The EFs values of the heavy metals in Lake Naivasha sediment were comparable with those of Taihu Lake, China as presented by Yuan *et al.* (2011). It was found out that the mean EF values for sediment in Lake Naivasha ranged from 0.3 to 3.7 while those of Taihu Lake were 0.4 – 3.7, respectively. Lake Naivasha, Taihu Lake are both freshwater lakes that are surrounded by agricultural activities and receives effluent from domestic wastewater.

The results show that the concentration of As, Fe Mn and Zn metals in most lake sites had been affected by anthropogenic sources while other metals were not. This phenomenon was observed from all pollution indices since the metals were classified as lowly contaminated to moderately contaminated classes depending on the various groupings of the indices. From the indices, it was possible to establish the source of heavy metal a finding that agreed with the report by Rabee *et al.* (2011) which indicated that CF, I_{geo} and EF indices help in distinguishing natural and anthropogenic input on heavy metals. With some metals being anthropogenically influenced, this suggests that there is need for conservation practices to be adopted within the basin.

4.4 Determining Areas to be Prioritized for Conservation Practices using SWAT Model

Spatial distribution for soils, slopes and land use/cover change of Lake Naivasha basin are presented. Further, sensitive parameters and results on calibration and validation of SWAT model are also presented. From the calibrated model, sediment load into Lake Naivasha, spatial distribution of sediment yield within Lake Naivasha basin over time and findings on impacts of conservation practices such as terraces and filter strips are given.

4.4.1 Characterization of Lake Naivasha Basin

The different soils within Lake Naivasha basin are presented in Fig 4.16. The predominant soils (World Reference Base classification) in the basin are Planosols (PLe), Phaeozems (PHh) and Andosols (ANm) covering approximately 34.78, 17.42 and 9.18% of the total area, respectively (Appendix 5, Table A5). Andosols have high potential for agricultural activity since they are highly fertile, easy to cultivate and encourages root penetration. On the other hand, Planosols are poorly drained and allow very little infiltration thus generating high volumes of surface runoff. With increase in surface runoff there are possibilities of increased erosion leading to high sediment yield from the basin. According to Arnold *et al.* (1998), the SWAT model uses the Hydrologic Soil Groups categorized as A, B C and D (A has the greatest infiltration potential while D soils have greatest runoff potential) in determining runoff potential of an area. Soils in Lake Naivasha basin fall under Hydrologic Soil Groups (HSG) B (25.9%), C (73.1%), and D (1%). Soil characteristics, coupled with other landscape factors, such as topography and land use/cover are useful in delineating Hydrological Response Units (HRUs) in SWAT (Arnold *et al.*, 1998).

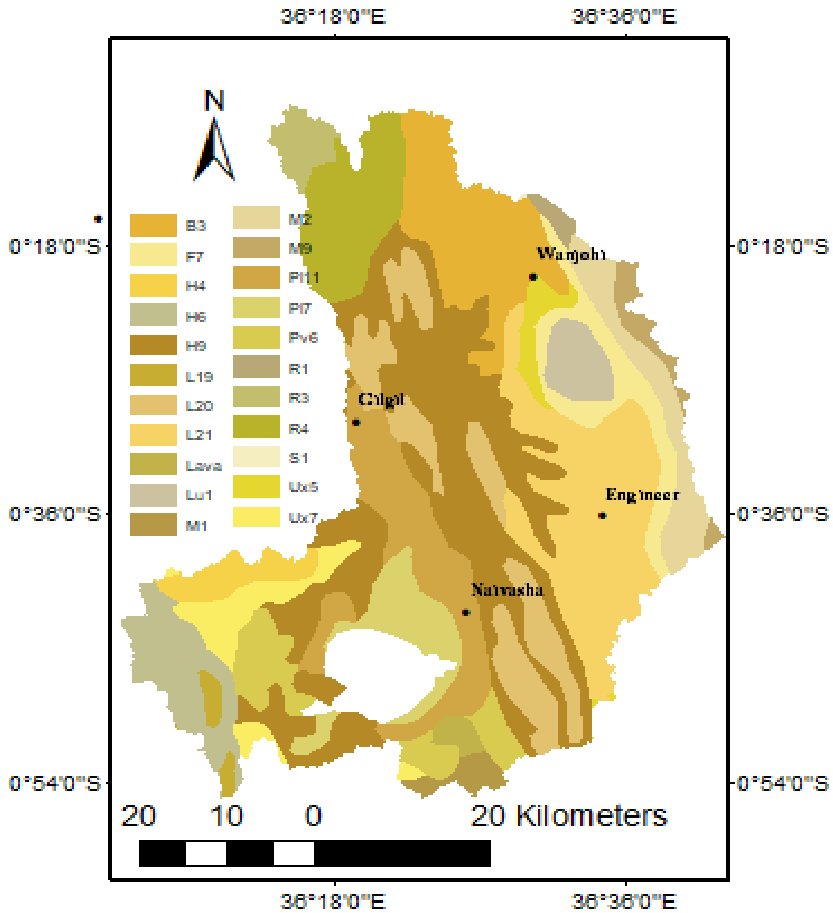


Figure 4.16: Spatial distribution of Soils within Lake Naivasha basin

Topography

Predominant slopes within Lake Naivasha basin were found to be between 0 and 20 % accounting for approximately 76% of the total area. Approximately 46 % and 30.7 % of the surface slopes within the basin, range between 0-10 % and 10-20 %, respectively. Towards Lake Naivasha, relatively flat slopes exist in the area. Steep slopes of above 30% occupy an area of about 6.7%. The steep slopes were found to be dominant in forested and escarpment regions within the basin. In SWAT model, topography is represented using DEM. The topography is useful in defining average slopes, slope length, and flow accumulation. Further, topographic relief of a basin influences stream flows, sediment and nutrient transports from sub basins to the stream reaches which is influenced by the slope length and degree, and the contributing area (Arnold *et al.*, 1998). From the results it was observed that agricultural lands within Lake Naivasha basin fall on slopes ranging from 0 - 30 %.

Land use and cover

The land use/cover situation of Lake Naivasha basin between 1972 – 2015 periods are presented in Figures 4.17 a, b, c and d. It was found that a significant amount of land use/cover changes occurred within the basin where land uses in various sub basins were found to change between 1972, 1984, 2000 and 2015.

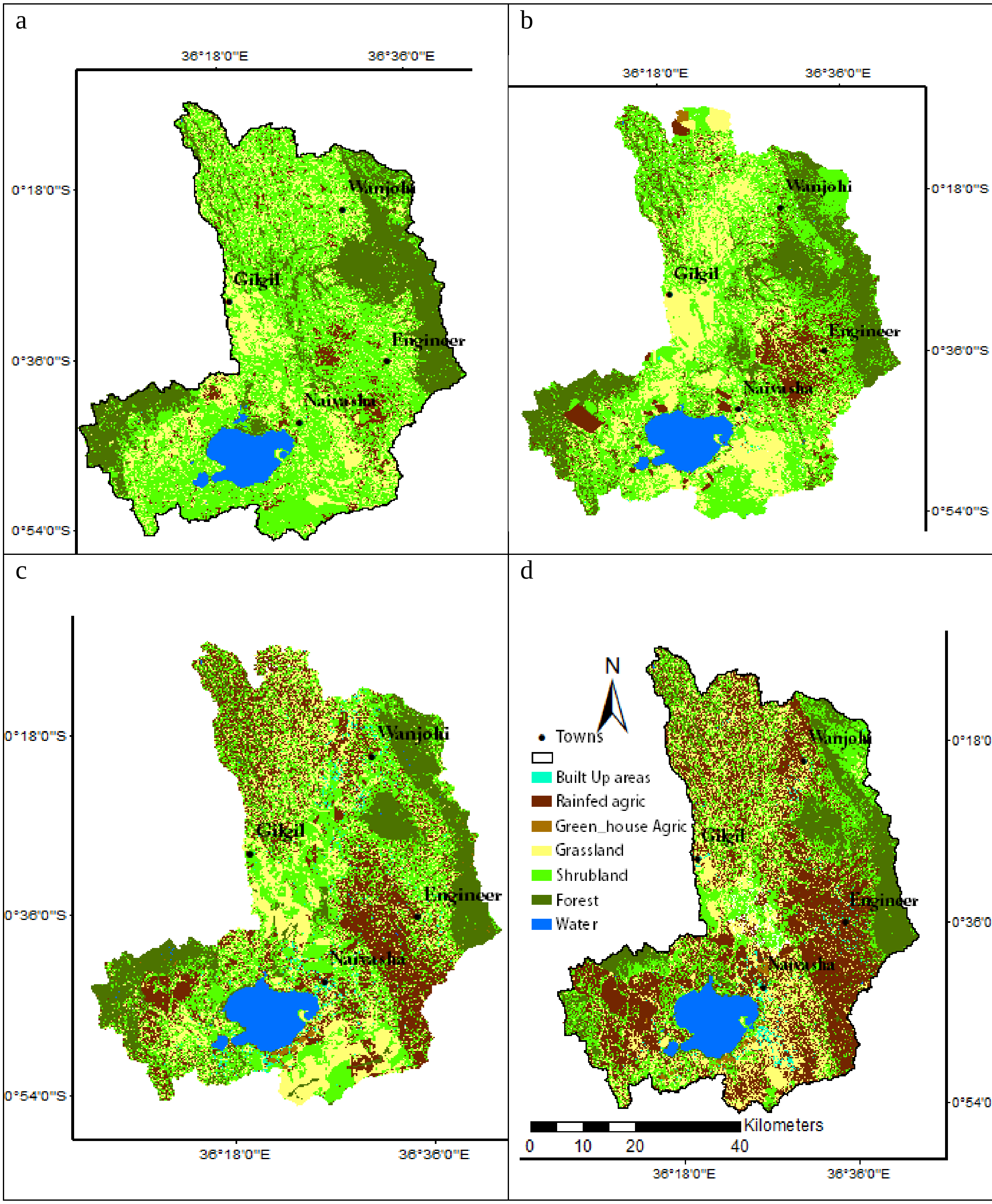


Figure 4.17: Land Use /cover maps of Lake Naivasha basin (a)1972 map (b) 1984 map (c) 2000 map and (d) 2015 map

The summary of land use/cover for Lake Naivasha basin between 1972 and 2015 is presented in Table 4.8

Table 4.8: Land use and SWAT classes for Lake Naivasha basin in 1972, 1984, 2000 and 2014

Land use/cover category	SWAT classification	% Land cover area within the basin			
		1972	1984	2000	2015
Built up area	URMD	0.0	0.1	0.98	1.50
Rain-fed Agriculture	AGRL	4.6	9.0	22.68	36.59
Green house Agric	AGRC	0.0	0.0	0.80	1.94
Grassland	RNGE	20.9	24.3	21.63	18.06
Shrubland	RNGB	49.0	41.3	31.53	23.60
Forest	FRST	21.2	20.9	18.00	13.64
Water	WATR	4.1	4.4	4.4	4.66
Total Area		100.0	100.0	100.0	100.00

The change of land use and land cover in Lake Naivasha basin especially the decline in forest areas and the increase of areas under agriculture and settlement activities was observed (Table 4.8 and Appendix 5, Figure A5). The forested area, shrubland and grassland were found to have reduced throughout the study period (between 1972 and 2015). The total forest, shrubland and grassland reduction between 1972 – 2015 were found to be 7.56, 25.4 and 2.84 %, respectively. Further, between 1972 – 1984, 1984 – 2000 and 2000 – 2015, the forest reduction was 0.3, 2.9 and 4.36 %, respectively (Table 4.8). On the other hand, agricultural land within Lake Naivasha basin was found to increase by 33.93 % between 1972 and 2015. From the analysis it can be deduced that most of the land was converted from natural land (forest, shrubland and grassland) to agricultural land use. This may result to variation of sediment yield within various sub basins over time thus impacting on Lake Naivasha sedimentation rate. Irrigated agriculture mainly supporting horticulture and floriculture in areas near the lake have been increasing (Figure 4.17). This finding agreed with those reported by Onywere *et al.* (2012) for a study conducted in Lake Naivasha basin, Kenya. Onywere *et al.* (2012) observed that agricultural land increased between 1985-2006 while the natural landcover was on the decline. In their study they found that small-scale farmlands were on the increase while large-scale farmlands decreased by about 27.6%. Within the same study grassland was reported to have reduced by 10.2% between 1985-2006. The current study found that adjacent to forested region, agriculture occupies most of the land. This indicates that there is need to have soil conservation measures adopted in the region. Dense agricultural activities have been observed in the upper and middle parts of the basin where crops such as potatoes, cabbages and carrots are grown.

Over the study period, a slight increase of water body was observed. Similarly, the extent of built-up areas was found to have increased between 1972 and 2015 (0.02 to 1.51 %), as shown in Table 4.8. Cases of agricultural lands and built up areas increasing while natural land is decreasing had been reported by Gashaw *et al.* (2018), Haile and Suryabhagavan (2018) and Lei and Zhu (2018). Changes in land use/cover have the potential to alter not only the quantity and rates of sedimentation but also the water quality in lakes, reservoirs and streams (Lexartza-Artza and Wainwright, 2011; Gyamfi *et al.*, 2016). Thus, to determine the impacts of land use change on Lake Naivasha sedimentation, the SWAT model was used. The classified images were coded to SWAT land use Land cover classification as given in Table 4.8. The combined Land use, Soils and slopes aided in generating the HRUs. A total of 71 sub basins and HRUs were determined since the dominant land use was assigned during HRU definition. This aided in assessing sediment yield from various HRUs after the SWAT model sensitivity analysis, calibration and validation.

4.4.2 SWAT Model Sensitivity Analysis

From sensitivity analysis, it was observed that two soil parameters namely; Soil Available Water Content (Sol AWC) and Soil hydraulic conductivity (Sol K) had greater impact on the model output. These parameters control the amount of water that may be retained in the soil profile further impacting on the water balance. An increase in Sol AWC resulted into a reduction of surface runoff, base flow and water yield. Kannan *et al.* (2007) also observed that all water balance components were sensitive to variation of Sol AWC. Since most of the soils in Lake Naivasha basin were multi-layered, the Sol K at the surface layer was reduced in order to lower the lateral flow. The reduction of soil hydraulic conductivity led to an increase in surface runoff since infiltration rate was lowered.

In addition, soil evaporation compensation factor (ESCO) was found to be a sensitive parameter which affected all components of the water balance. A decrease in ESCO results to reduction of surface runoff, base flow and water yield. On the other hand, a decrease in ESCO lead to an increase in evapotranspiration (Sang *et al.*, 2015). With reduction of ESCO, the model tends to extract more of the evaporative demand from lower layers (Kannan *et al.*, 2007). ESCO accounts for effects of capillary action by adjusting the depth distribution of evaporation from the soil (Venkatesh *et al.*, 2018). On the other hand, the Curve Number (CN) was found to be sensitive to annual flows. Other parameters that impacted on the stream flows included the average slope steepness. The average slope steepness was found to largely impact on water balance and sediment yield.

The groundwater parameters that were found to be sensitive were; GWQMN, GW_Delay, Recharge DP, GW_Revap and Alpha Bf. The base flow factor is useful in describing the response time of ground water to changes in recharge. The threshold water depth in shallow aquifer (GWQMN) impacted on ground water and base flow component of the water balance. It was observed that lower values of GWQMN increased the base and stream flows. In SWAT model, the base flow enters the stream only if the depth of water in shallow aquifer exceeds GWQMN (Kannan *et al.*, 2007). It was also observed that the slope and average slope length SLSUBBSN are sensitive to both water balance and sediment. Other model parameters that were found to be highly sensitive to sediment included linear re-entrainment parameter for channel sediment routing (SPCON), and exponent of re-entrainment parameter for channel sediment routing (SPEXP). These most sensitive parameters were useful during the SWAT model calibration and validation. The importance of channel processes to sediment yield were also highlighted by Kumar *et al.* (2015) and Moriasi *et al.* (2011) who found out that SPCON and SPEX greatly impacted on sediment simulation.

4.4.3 SWAT Model Calibration and Validation Results

The calibration and validation improved the agreement between the observed and simulated annual discharges. The calibrated hydrologic parameters and the corresponding range of values for the parameters were CN (47-77), AWC (0.08-0.49), ESCO (0.35-0.88) and SolK (0.3-14). For the ground water parameters, calibrated hydrologic parameters included; GWQWN (1000-2500), GW_Revap (0.05-2.5) and RCHRG_DP (0.08-0.25).

The statistical performance indices during calibration and validation of SWAT model for Lake Naivasha basin are presented in Table 4.9. The PBIAS, was found to range between -2.086 and +6.652 during calibration. Further, considering validation period, the PBIAS was found to range between -24.74 and 28.24.

Table 4.9: Performance indicators on hydrological SWAT calibration and validation for various gauges within Lake Naivasha basin

	Gauge No	Flows (m^3/s)		Performance Indicators		
		Observed	**Simulated**	**NSE**	**r^2**	**PBIAS**
Calibration	2GA03	0.33	0.329	0.63	0.75	-0.24
	2GB01	4.792	4.689	0.69	0.79	-2.09
	2GB07	1.166	1.243	0.67	0.71	6.65
	2GC04	2.264	2.251	0.83	0.83	-0.59
Validation	2GA03	0.781	0.532	0.3	0.11	-24.74
	2GB01	-	-	-	-	-
	2GB07	1.029	1.432	0.37	0.49	25.24
	2GC04	3.389	3.185	0.54	0.58	-6.03

During the validation periods the NSE and r2 values for gauge 2GA03 and 2GB07 were found to be less than 0.5. This was because of large data gaps that was on the observed stream flow data over the validation period. It was not possible to change the validation period because the data of the said period were up to 1992. Also, for 2GB01 gauge there was no observed data over the calibration period. From the results, it was found that the model performance was satisfactory based on the statistical criterion as reported by Moriasi *et al.* (2007). According to Betrie *et al.* (2011) and Moriasi *et al.* (2007), the model simulation is considered satisfactory if NSE $\geq$ 0.5, $R^2 \geq 0.50$ and PBIAS = ±25% for flow and NSE $\geq$ 0.5, RSR $\leq$ 0.70 and PBIAS = ± 55% for sediment are achieved during calibration and validation processes.

On the other hand, the calibrated sediment parameters included the linear parameter for calculating the maximum amount of sediment that can be re-entrained during channel sediment routing (SPCON) which ranged between 0.0003–0.0004. Also, the exponent parameter for calculating sediment re-entrained in channel sediment routing (SPEX) was a calibration parameter and was found to range between 1.2 and 1.4. The sediment calibration on annual time step was found to yield a PBIAS value of -4.5, 3.4 and -1.2%

for the periods between 1978–1992, 1993–2004 and 2005–2015, respectively. On the other hand, the NSE for the same study period were found to be a range of -0.05 – 0.132 while the r^2 values ranged between 0.1 and 0.35. Considering the PBIAS, SWAT model was found to be satisfactory in sediment simulation since according to Moriasi *et al.* (2007), PBIAS values of ≤ 55% for sediment simulation indicates a good performance. With parameter transfer for the periods between 1993-2004 and 2005-2015, it was observed that the SWAT model performance was satisfactory. Sang *et al.* (2015) showed that with parameter transfer, over a similar period, (short-term or long-term) the performance of SWAT model improved. Hence, the calibrated model was used to identify major sediment source areas within the Lake Naivasha basin.

4.4.4 Lake Naivasha Basin Sediment Yield

From the calibrated model it was found that sediment hotspots within the basin have been changing over the 1981 – 2015 period. This was evidenced by the observation made that simulated annual sediment yield of various sub-basins in Lake Naivasha basin varied between 1981 – 1992, 1993 – 2004 and 2005 – 2015 periods (Figures 4.18 a, b and c). The sediment classification criteria used in this study was adopted from Betrie *et al.* (2011) and Gathagu *et al.* (2017). The simulated sediment yield from various sub basins, between 1981-1992, 1993-2004 and 2005-2015 periods, ranged from 0.00 – 14. 95, 0.00 – 22.94 and 0.00 and 47.16 tons/ha/yr, respectively. High sediment yield was observed in sub basins with extensive agricultural activities. Odhiambo and Ricker (2012) also reported that agricultural fields contributed a high sediment load into Lake Anna, in Virginia US. The variation of sediment yield for all sub basins over the study period is presented in Appendix 6.

It was found out that SWAT model closely predicted sediment yield within Lake Naivasha basin. The results of this study closely agreed with those reported by Njogu and Kitheka (2017) who observed that SWAT model was useful in predicting sediment yield in a highly human-impacted tropical catchment.

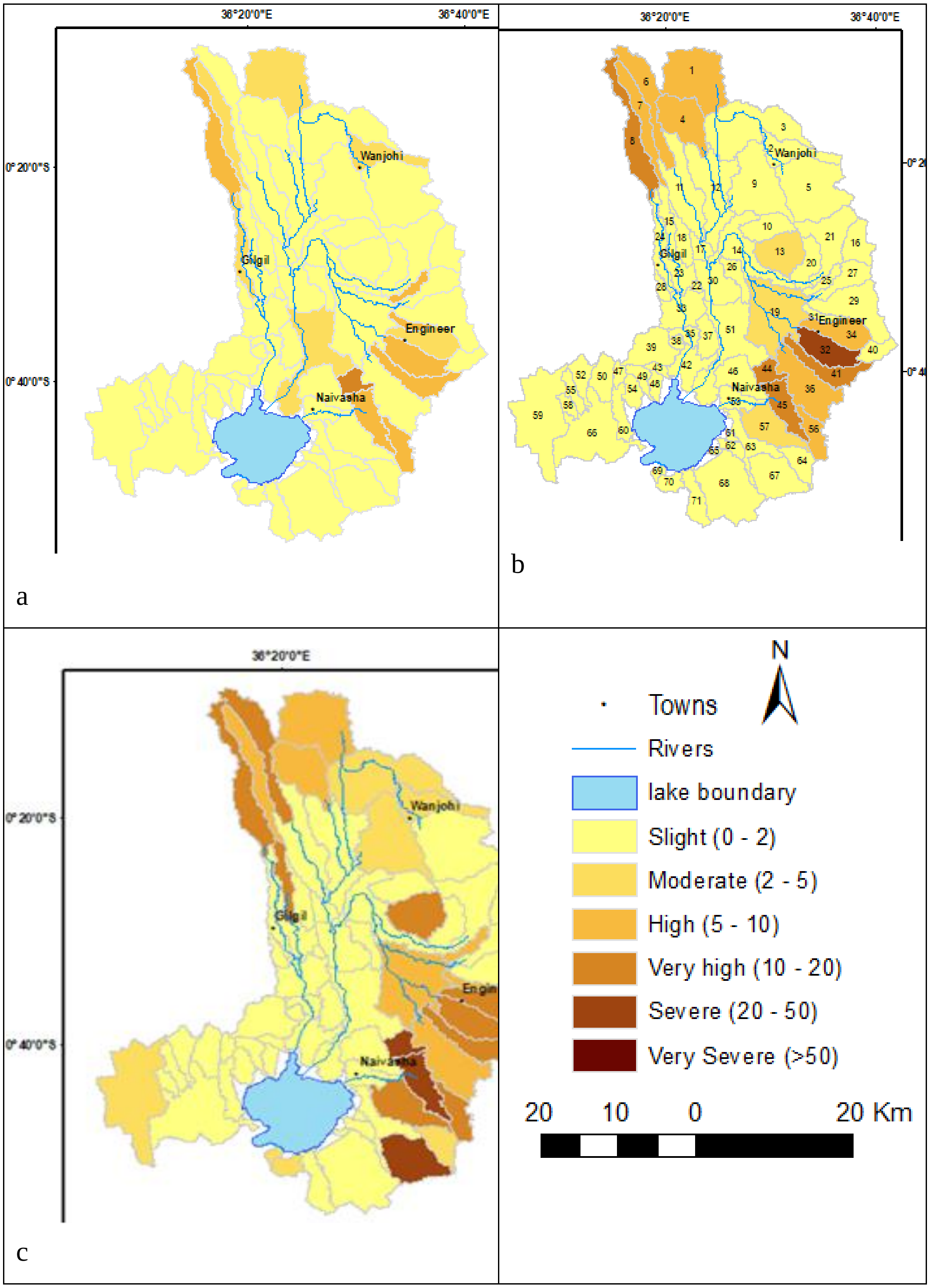


Figure 4.18: Lake Naivasha basin Sediment yield (t/ha/yr) from various sub basins between (a) 1981-1992, (b) 1993-2004 and (c) 2005 -2015 periods

During 1981 – 1992, 1993 – 2004 and 2005 – 2015 periods, the number of sub basins that were found to fall in high to very severe sediment yield class were 9, 11 and 16 sub basins, respectively (Table 4.10).

Table 4.10: Sub basins with sediment yield ≥ 5 ton/ha/yr between 1981 – 1992, 1993 – 2004 and 2005 – 2015 periods

Rank	Period					
	1981 - 1992		1993 - 2004		2005 - 2015	
	Sub basin	Sed yield (t/ha)	Sub basin	Sed yield (t/ha)	Sub basin	Sed yield (t/ha)
1	44	14.95	32	22.94	44	47.16
2	34	12.26	44	17.67	67	22.45
3	32	8.75	41	12.95	45	22.26
4	41	7.80	45	10.63	57	20.73
5	45	6.73	6	10.39	41	17.75
6	31	6.72	8	10.08	32	17.06
7	25	6.59	56	9.18	56	16.70
8	56	5.97	4	6.45	13	16.45
9	8	5.37	7	5.64	15	15.22
10			36	5.35	7	14.79
11			1	5.19	8	13.57
12					31	9.52
13					25	9.27
14					36	7.67
15					4	6.16
16					1	5.76

This indicates that in the recent years, there has been an increase in sediment source critical areas within Lake Naivasha basin. According to Betrie *et al.* (2011) and Gathagu *et al.* (2017), sediment yield ≥5 tons/ha/yr is classified to fall within high and very severe classes. Kumar *et al.* (2015) also reported that sub basins with soil loss ≥5 ton/ha/yr are critical and should be prioritized for conservation purposes.

Anthropogenic activities especially intensive agriculture was found to be dominant in most sub basins that were classified as critical areas. Swallow *et al.* (2009) reported similar findings where high sediment yield was observed in regions impacted by anthropogenic activities especially agricultural practices. The variation in sediment yield from various sub basins was found to impact on sediment loading into Lake Naivasha over time. According to Licciardello *et al.* (2017), the change in land uses especially of forests and rangelands to agricultural land leads to an increase in soil erosion further impacting on sediment load into lakes and reservoirs.

4.4.5 Sediment Load into Lake Naivasha

Figure 4.19 presents the cumulative annual observed and simulated sediment load into Lake Naivasha. The cumulative annual sediment load was used since natural lakes and reservoirs are impacted by sediments in the form of cumulative loss of storage capacity.

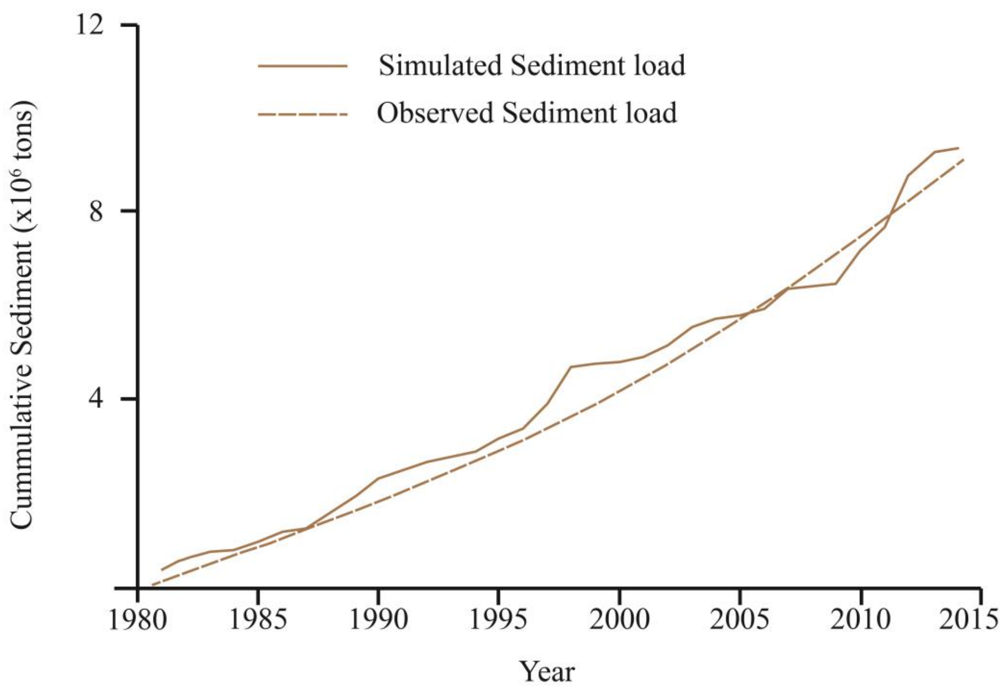


Figure 4. 19: Cumulated annual observed and SWAT simulated sediment load into Lake Naivasha between 1981 – 2014

From Figure 4.19, it was observed that the cumulated observed sediment closely agreed with the cumulated simulated sediment load into Lake Naivasha with variations experienced in the periods with prevailing extreme weather condition around 1997-1998 (floods) and 2008-2009 (drought). The impacts of weather conditions on sediment load was not easily visible in the observed data. This is because sediment core sampling was based on visual stratigraphic changes thus resulting to interpolation of the dated layers.

The cumulative observed and simulated sediment load to Lake Naivasha between 1981-1992, 1993-2004 and 2005-2015 is presented in Table 4.11. It was observed that sediment load into the lake has been increasing in the recent past.

Table 4.11: Total cumulative sediment Load into lake Naivasha between 1981 – 1992, 1993 – 2004 and 2005 - 2015 period

Period	**Total Observed sediment (x10^6 tons)**	**Total Simulated sediment (x10^6 tons)**
1981 - 1992	1.53	1.6
1993 - 2004	3.13	3.02
2005 - 2015	3.58	3.62

The increase of sediment load into Lake Naivasha is also shown by the gradient of the cumulative sediment (Figure 4.20). It was found that the gradient for 2005 - 2015 is the highest while gradient for 1981 – 1992 was the lowest.

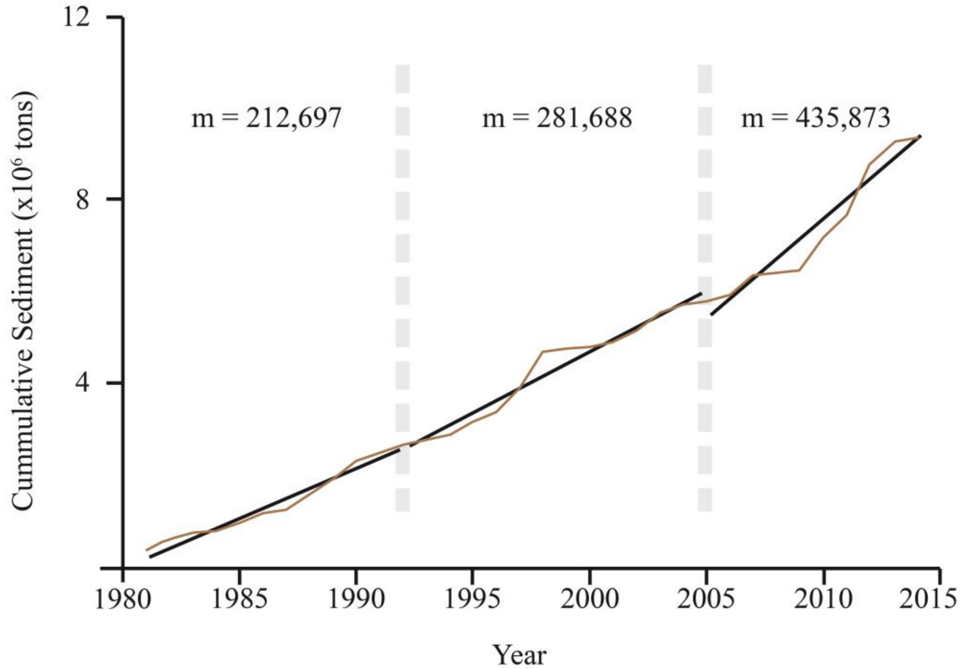


Figure 4.20: Slope of cumulative simulated sediment load in Lake Naivasha, Kenya for 1981 – 1992, 1993 – 2004 and 2005 – 2015 period

The increase in sediment load could mainly be attributed to anthropogenic influence. For instance, it was observed that there has been an increase of agricultural land within Lake Naivasha basin. According to Pilgrim *et al.* (2015) changes in land cover especially reduction of vegetation cover within a basin results in billions of tons of extra sediment being deposited in streams, lakes and reservoirs. A close agreement of observed and SWAT simulated results on sediment load into a lake/reservoir had also been reported by Feyissa (2016) for Gigel Gibe-1. The author reported that the observed and simulated sediment load into Gigel Gibe reservoir over the past 20 years (1990-2010) was 3,442,064 tons and 3,446,300 tons, respectively.

Comparing APS generated and SWAT simulated data for the past 20 years (between 1996 – 2016), the total cumulated APS and total cumulated simulated sediment load was found to be 6.05 x10^6 and 6.17 x10^6 tons, respectively. The findings of this study agreed with those of Moriasi *et al.* (2011) where APS collected data and SWAT simulated data were comparable.

From this study, it was found that the average annual reservoir sedimentation load into Lake Naivasha obtained from multifrequency APS served as a potential source of data for calibrating SWAT model. Similar findings were reported by Moriasi *et al.* (2011) in a study conducted on crowder Lake. It was found that the APS data aided in understanding short- and long-term annual sedimentation status. This is especially necessary in most ungauged watersheds. Long-term annual average sedimentation rates obtained from bathymetric survey using APS were useful in calibrating and validating the SWAT model for sedimentation studies in ungauged watersheds (Moriasi *et al.*, 2011). The use of APS data in calibrating SWAT model was found to be useful in improving the model output on simulated sediments further aiding in assessment of impacts of conservation practices within the basin. According to Palmieri *et al.* (2001) sediment load reduction and benefits accrued from lakes can be optimized by adopting different sediment management strategies. The impacts of best management practices can be well understood through model simulation before they are implemented within the basin.

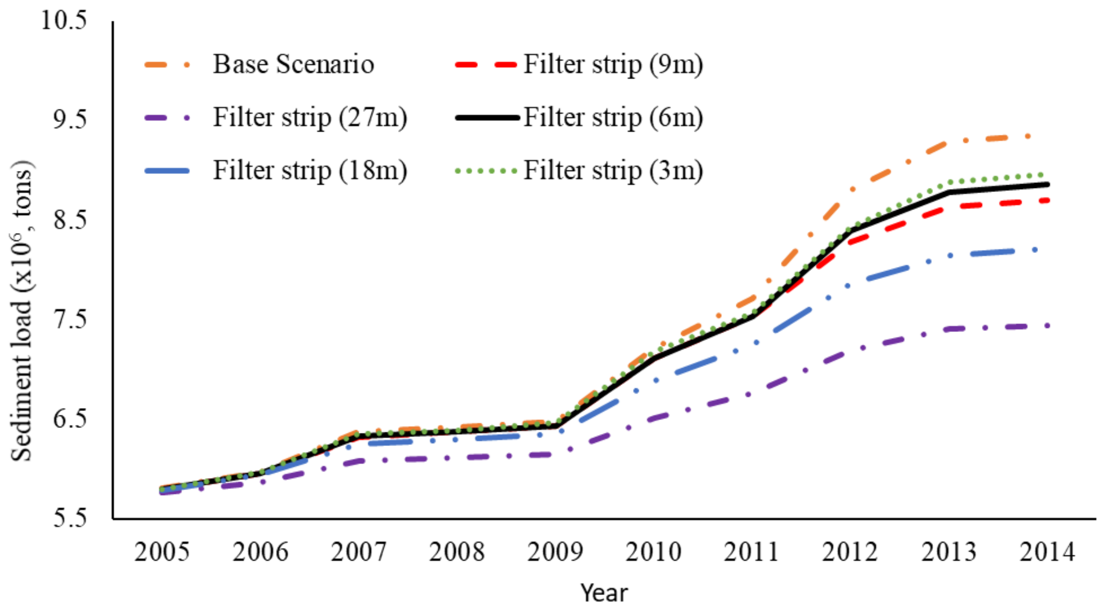
4.4.6 Impacts of Conservation Practices on Sediment Yield and Cumulative Sediment Load into Lake Naivasha

From SWAT model simulation, it was observed that reduction of sediment yield from critical sub basins and or all agricultural land using either terrace or filter strips (riparian buffer), subsequently lowered the cumulative sediment load into Lake Naivasha. Further, it was observed that implementation of a conservation practice had a higher reduction on sediment yield from the target areas compared to reduction of cumulative sediment load into Lake Naivasha.

With implementation of filter strips, it was observed that sediment yield reduction in the target areas would reduce up to 45.31% and 73.56% for critical areas and agricultural sub basins, respectively. On the other hand, filter strips implementation within critical areas sub basins, reduced cumulative sediment load into Lake Naivasha from 3.13 x 10^6 to 1.71 x 10^6 tons. Further, from simulation of filter strips being implemented in all sub basins with agricultural activities within the basin, it was found that cumulative sediment load would reduce from 3.13 x 10^6 and 0.83 x 10^6 tons depending on the width of the filters.

Increase in filter strip width (3, 6, 9, 18 and 27m), resulted to a decrease in cumulative sediment load into Lake Naivasha as given in Figures 4.21 a and b. The reduction of cumulative sediment load into Lake Naivasha was found to be 4.3, 5.4, 7.1, 12.3 and 20.5% if filter strips of 3, 6, 9, 18 and 27m width are implemented in all sub basins identified as critical areas. On the other hand, filter strips of the same width were simulated in all sub basins with agricultural activities, cumulative sediment loading into Lake Naivasha reduced by 5.5, 8.07, 10.0, 19.3, and 30.0%, respectively. From this study, it was observed that even filter strips of 3m would have an impact on sediment loading into Lake Naivasha. Ogweno *et al.* (2010) reported a 17% sediment load reduction into Lake Naivasha if filter strips of 5m width would be adopted on the priority sub basins. This value was higher compared to the finding from this study, where a 6m filter strip would result to 5.4% sediment load reduction. The difference could be because Ogweno *et al.* (2010) had not calibrated sediment process within SWAT model. This would then result to overestimation of impacts of conservation practices on sediment reduction. Sediment calibration on hydrologic models significantly reduces uncertainty of simulated sediment outputs and impacts of conservation practices (Moriasi *et al.*, 2011).

a)



b)

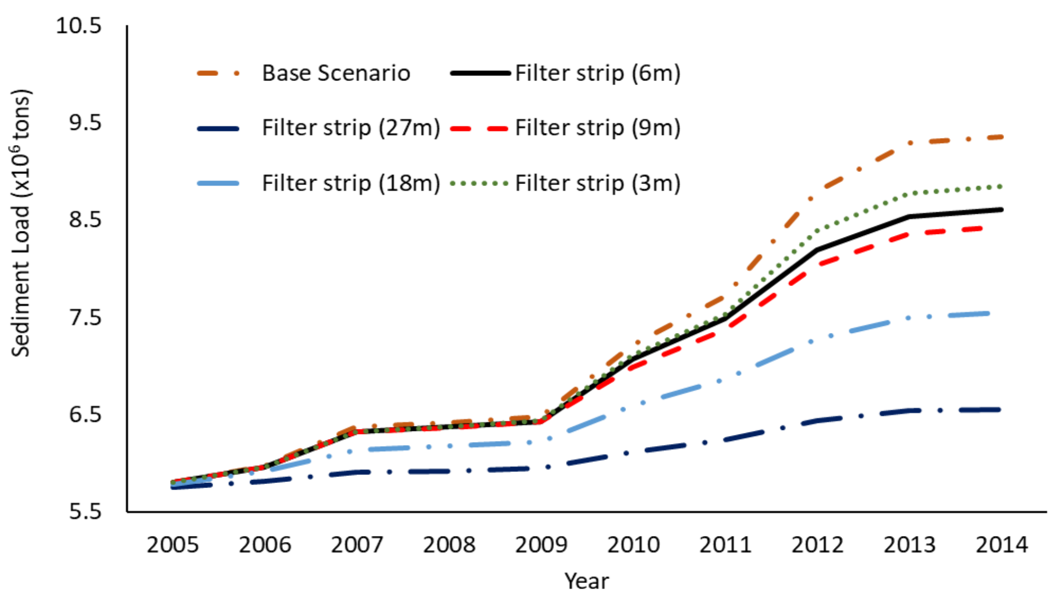


Figure 4.21: Impacts of implementing filter strips on cumulative sediment load into Lake Naivasha (a) Filter strips implemented on all critical areas and (b) Filter strips implemented on all agricultural land within the basin

According to Yuan *et al.* (2009) the width of a buffer is important in filtering agricultural runoff where wider buffers tend to trap more sediment. The authors reported that riparian buffers greater than 6 m are effective and reliable in reducing sediment load from any situation. Gathagu *et al.* (2018) demonstrated that the width of filter significantly affects sediment yield. Gathagu *et al.* (2018) reported that a reduction of 36% and 46% sediment yield would be achieved from the 3 and 6 m wide filter strips, respectively. According to guidelines on riparian land in Kenya, riparian area should be a range of 2 to 36 m and should be left on both sides from the highest water mark during peak flows. Thus, if these guidelines are followed, the amount of sediment reaching waterbodies including Lake Naivasha would be reduced. However, it was observed that impacts of filter strips above 30 m could not be simulated using the SWAT model. This could be attributed to the trapping efficiency equation (Neitsch *et al.*, 2005; Arabi *et al.*, 2008) used in the SWAT model (Equation 4.1). From the equation, it was observed that the simulation of 30 m and above would result to trap efficiency of unity.

$$trap_{eff\_sed} = 0.367 \times FILTERW^{0.2967}$$

(4.1)

where,

$trap_{eff\_sed}$ = trapping efficiency of the sediments

$FILTERW$ = the width of the filter strip (m)

The findings from this study closely agreed with those found by Kumar *et al.* (2015) for Komar and Panchet catchments. Kumar *et al.* (2015) reported that simulated sediment before management practices were 1.19 and 4.32 Mm^3/yr for Konar and Panchet reservoirs, respectively. The authors reported that the sediment values reduced to 1.04 and 2.13 Mm^3/yr which translates to 12.6 and 50.64 % reduction after adoption of management strategies in the catchments.

From the findings of this study, it was observed that a greater impact of conservation practices would be on sediment yield reduction at the intervention areas compared to lowering of cumulative sediment load into Lake Naivasha. Thus, from the results, these management practice would be effective in retaining fertile soils within the land but their impact on reduction in sediment load into Lake Naivasha would be lower. According to Arabi *et al.* (2008) and Betrie *et al.* (2011), the filter strips reduced sedimentation of reservoirs since they filter the runoff and trap the sediment from a given area. Also, filter strips reduce overland flow, slope length and sheet erosion thus reducing sediment transport capacity. For successful implementation of filter strips, direct benefits from strips would encourage farmers to preserve and maintain them. In implementation of filter strips, vegetation to be used can be selected to have multiple functions. For instance, if trees and grass are used for filter strips, the grass may be used as feed to livestock while trees could also be useful to farmers. The use of filter strips would aid in sedimentation and water body pollution control. However, the impact on sediment load reduction into Lake Naivasha could be improved by adopting more than one conservation practice for instance the use of filter strips and implementation of terraces.

Introduction of terraces in all agricultural land was found to have a higher reduction of sediment load compared to introducing terraces on sub basins that were identified to be critical areas (Figure 4.22). It was found that the introduction of terraces on all agricultural lands and on the observed critical areas within the basin reduced sediment load by 27.2 and 18.8%, respectively. On the other hand, with implementation of terraces, sediment yield reduction on the target sub basins was found to be 40.2 and 65.5%, respectively.

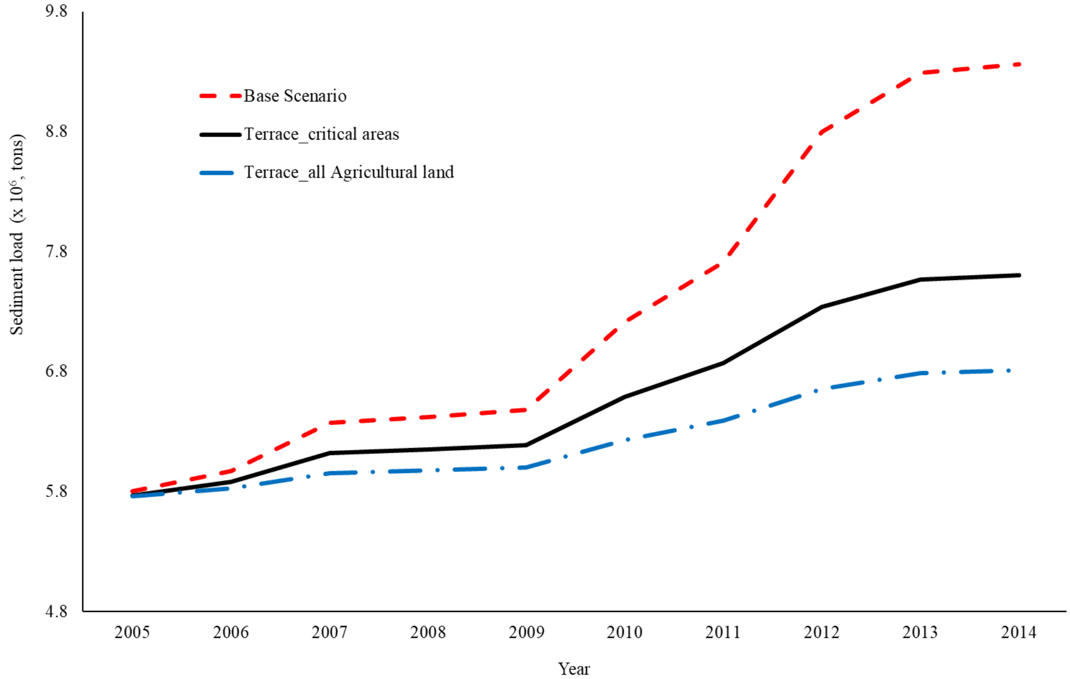


Figure 4.22: Impacts of terraces on sediment load into Lake Naivasha (Terraces implemented on all hotspots and agricultural land within the basin)

The impact at the sub basins level was also found to be higher than that of cumulative sediment load reduction. These results are comparable with those reported by Betrie *et al.* (2011) who observed that terraces would reduce sediment load by about 41% at Blue Nile Basin estimated using SWAT model. Lemann *et al.* (2016) also found that, implementation of conservation measures within farm levels resulted to a high reduction of sediment yield than at catchment level in a study conducted in upper Blue Nile catchment, Ethiopia. Further, effectiveness of terraces on reduction of sediment load into Lake Naivasha was comparable with those reported by Ayala *et al.* (2017). Ayala *et al.* (2017) reported that sediment from Hunde Lafto watershed of Upper Wabi Shebelle Basin reduced by 17.3% - 80.6% with introduction of terraces. Successful simulation of impacts of terraces in sediment load reduction using the SWAT model were reported by Arabi *et al.* (2008), Mwangi (2011) and Briak *et al.* (2019). Terraces are also effective in reducing diffuse pollution from agricultural lands (Santhi *et al.*, 2006; Arabi *et al.*, 2008). With use of terraces the slope length factors for various HRUs and sub basins reduces resulting in reduction of overland flow, soil detachment and transportation. Mwangi *et al.* (2015) found out that sediment yield reduction at sub basins level in Sasumua watershed would range between 0.0 to 32.5 % with adoption of various conservation practices. The authors advocated for comprehensive sediment calibration to improve on the model output. However, in this study, sediment calibration was comprehensive over a long period (1981-2016) by use of APS sediment data.

In assessing the impacts of conservation practices on sediment load reduction into Lake Naivasha, it was observed that intervention on all sub basins having agricultural practices had a greater impact. These findings agreed with those reported by Kokpinar *et al.* (2015) for Tungabhadra Reservoir which indicated that through priority treatment of the catchment areas, it was possible to reduce the sediment load and sedimentation rate. It was found that implementation of filter strips and terraces in Lake Naivasha basin would result to 20.4 to 59.3% reduction of phosphorous in sediment. This agreed with the findings by Yuan *et al.* (2009) who reported that conservation practices not only reduce sediment loading into a water body, but also lower the nutrients and other pollutants getting into the waterbody. In this study, the SWAT model was also used successfully to quantify sediment-loss reductions as a basis for conservation practice.

However, according to a study by Palmieri *et al.* (2003) for large basins, the authors reported that sediment load reduction into water bodies after adoption of management practices in a small part of the basin was low. Verstraeten *et al.* (2006) reported that the impact of filter strips at a catchment scale was much less pronounced compared to reduction of sediment load at farm level. The authors found that sediment load into the reservoir would reduce by 17%. The use of filter strips on some parts of watershed, would result to by passing of sediment through ditches and roads. This lowers the effectiveness of the filter strips when assessed at a catchment scale. These findings agreed with those reported by Kidane and Alemu (2015) who assessed the effect of upstream land use practices on soil erosion and Sedimentation in the Upper Blue Nile

Basin, Ethiopia and found that implementation of conservation practices reduced sediment load by 11 % only. The lower reduction of sediment load into waterbodies was attributed to the fact that implementation of conservation practice targeted a smaller area within the basin.

CHAPTER FIVE

CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

i. The bathymetric survey and geochronological analysis of sediment cores aided in establishing the volume and sedimentation status of Lake Naivasha. This was based on multifrequency Acoustic Profiling System (APS) data collected between July and October 2016. It was observed that there has been an increase on sedimentation of lake Naivasha recently. This was depicted by the bathymetry survey finding and sediment dating. From the analysis, the lake was found to have mean depth, volume and surface area of 4.68 m, 722 $x10^6$ m^3 and 154.17 x 10^6 m^2, respectively. Considering water level at 1889 m, it was found that from 1983 and 2016 survey, the maximum water depths in the Main Lake (hippo point), Crescent Island Lake and Lake Oloiden reduced by 2 m, 0.6 m and 1.75 m, respectively. The change in depth was attributed to anthropogenic activities and environmental changes around the lake and this was counterchecked using dated sediment cores. The ^{137}Cs was successfully used as an independent time marker and aided in establishing depth age relationship using excess ^{210}Pb. A higher sedimentation rate was observed towards the inflow part of the lake. This was supported by the findings that the average mass sedimentation rate at Malewa (near the inflow) and Crescent sites of Lake Naivasha was 0.40 ± 0.04 and 0.26 ± 0.15 $g/cm^2/yr$, respectively. It was also found that, in the past 20 years (1996 – 2016) and 50 years (1966 – 2016), the maximum sediment thicknesses in the lake were 0.55 m and 1.9 m, respectively. The average sediment thicknesses within the same period were 0.25 and 0.56 m, respectively. The total volumes of accumulated sediments were found to be 36.35 x 10^6 and 53.38 x 10^6 m^3, respectively. This translated to sediment accumulation rates of 1.18 cm/yr and 1.10 cm/yr between 1996 – 2016 and 1966 – 1996, respectively. Two preferential sediment accumulation zones in Lake Naivasha, namely Hippo point and Crescent Island Lake were identified from the study. These zones corresponded with the deepest part of the Lake Naivasha. In this study, the combination of APS and sediment dating was also found useful in determining the sedimentation status of Lake Naivasha, while sediment cores were useful in geochronological and later geochemical analysis.

ii. Lake Naivasha sediment quality had been impacted by human activities. The geochemical analysis of Lake Naivasha sediments showed that, over the past 50 years, Al, As, Cr, Cu, Fe, Mn, Ni, Pb and Zn average concentration was 44457, 13.1, 15.0, 15.0, 46566, 1233, 14.6, 7.00 and 178.3 mg/kg, respectively. Multivariate analysis indicated that Al, Cr, Pb, Cu and Ni availability in the sediments was mainly due to geogenic control while availability of Zn, Mn, As, and Fe is controlled by both geogenic and anthropogenic sources. Further, Contamination Factor (CF) and geoaccumulation Index (I_{geo}) indices showed the sediment contamination classes ranged from uncontaminated to moderately contaminated. The average value Enrichment Factor (EF) index ranged between 0.3 and 3.2, with the highest EFs observed for Zn and Mn metals. The sediments were found to fall in no enrichment class for Cr, Cu, Ni and Pb metals while a minor to moderate enrichment class was observed for As, Mn, Fe and Zn metal. These indices showed that pollution increases on upper section of the cores compared to lower layers. The vertical variation of sediment quality along the core, as shown by the various indices, indicate a recent increase of anthropogenic activities within the basin. Thus, the quality of sediment along the core is useful in understanding variations of sediment input and sources over time.

iii. Simulation of cumulative sediment load into Lake Naivasha using SWAT showed spatial and temporal variation of critical areas that could be prioritized for conservation. Adoption of conservation practices within the basin would be useful in reducing sediment yield and load into Lake Naivasha. The model was successfully calibrated and validated using streamflow data from existing Regular Gauging Station (RGS) and average annual sedimentation data derived multifrequency APS. The simulation studies showed that between 1981-1992, 1993-2004 and 2005-2015 the number of sub basins that were identified as critical areas (sediment yield ≥ 5 t/ha/yr) were 9, 11 and 16 sub basins, respectively. The increase in number of sub basin identified as critical areas was mainly attributed to land use and cover changes within Lake Naivasha basin. In the basin, it was observed that between 1972 – 2016 the natural land (Forests, shrub and grassland) reduced by 35.8% while agricultural land increased by 33.9%. The calibrated model further aided in understanding the plausible impacts of conservation practices on sediment loading into Lake Naivasha. Scenario simulation showed that adoption of filter strips and terraces would reduce cumulative sediment load into Lake Naivasha by up to about 30% and 27%, respectively. In the two management scenarios, (i.e. use of filter strips and terraces), it was observed that if sub basins with agricultural practices are targeted a higher percent (up to 11%) reduction of cumulative sediment load into Lake Naivasha would be achieved compared to targeting critical areas only.

iv. Bathymetric survey, geochronological and geochemical analysis of sediment cores, and hydrological models is useful in identifying critical areas and quantifying the impacts of plausible interventions. Also, with the use of multifrequency APS in combination with dated sediment cores, it is possible to assess the sedimentation status of a natural lake that has no known pre-impoundment layer like reservoirs. This approach provides a solution for studying lakes and reservoirs where no prior comparable bathymetric surveys record exist.

5.2 Recommendations

i. The findings on Lake volume and current bathymetric condition of Lake Naivasha can be adopted to inform on policy for lake water abstraction. Also, the rates of lake sediment and sediment load obtained in this study can contribute toward understanding the impact of conservation practices adopted within the basin. This would further inform the managers on areas to be prioritized within the basin.

ii. From this study, further research be conducted to improve on the understanding of Lake Naivasha sedimentation and establishing areas that should be prioritized for conservation. For instance, grab samples for bank erosion studies can be collected along the various stream channels. This would improve on understanding sediment transportation from the source sub basins within the basin.
iii. In using pollution indices to investigate sediment sources, the background values of the elements under investigation are required. Thus, there is need of establishing regional background values other than relying on global estimated data. This would involve detailed soil mapping and analysis within the basin. Agrochemical presence in the sediments should also be determined
iv. From soil samples collected finger printing for instance isotope analysis should be conducted. This would improve on results of sediment budget within the basin. The combined stream channel characterization, soil fingerprinting would improve and further lower uncertainty from the SWAT model or any other hydrological model simulation.

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APPENDICES

Appendix I: Transects and tie lines followed during bathymetric survey

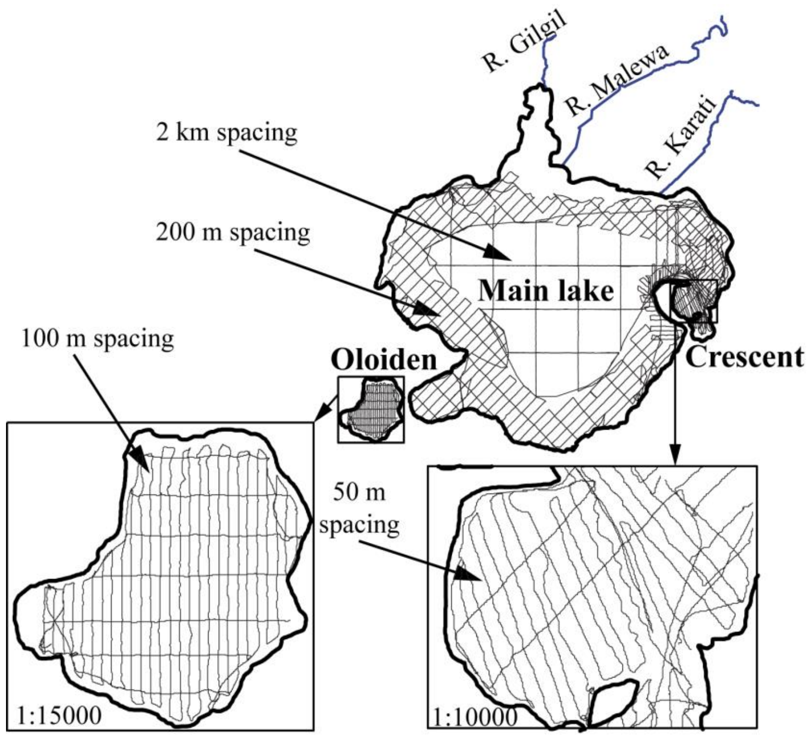


Figure showing the transects, tie lines and their spacings as used during bathymetry survey in Lake Naivasha and Lake Oloiden

Appendix Ib: Distribution of sediment core sampling sites in Lake Naivasha

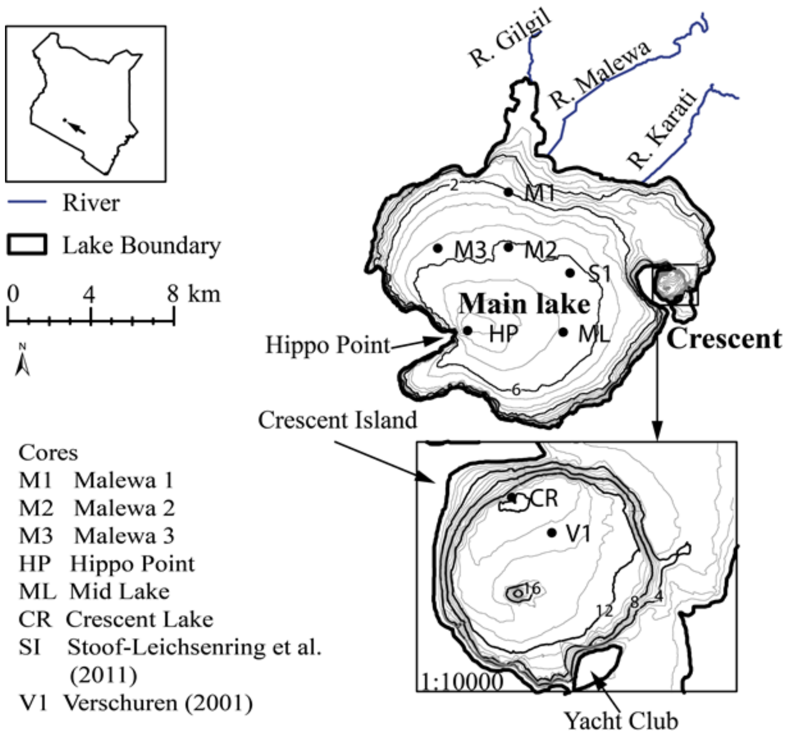


Figure on Location of Lake Naivasha, Kenya showing distribution of core sampling points and other dated cores S1 (Stoof-Leichsenring *et al.*, 2011) with V1 by Verschuren (2001)

Table on Sediment core information and location in Lake Naivasha

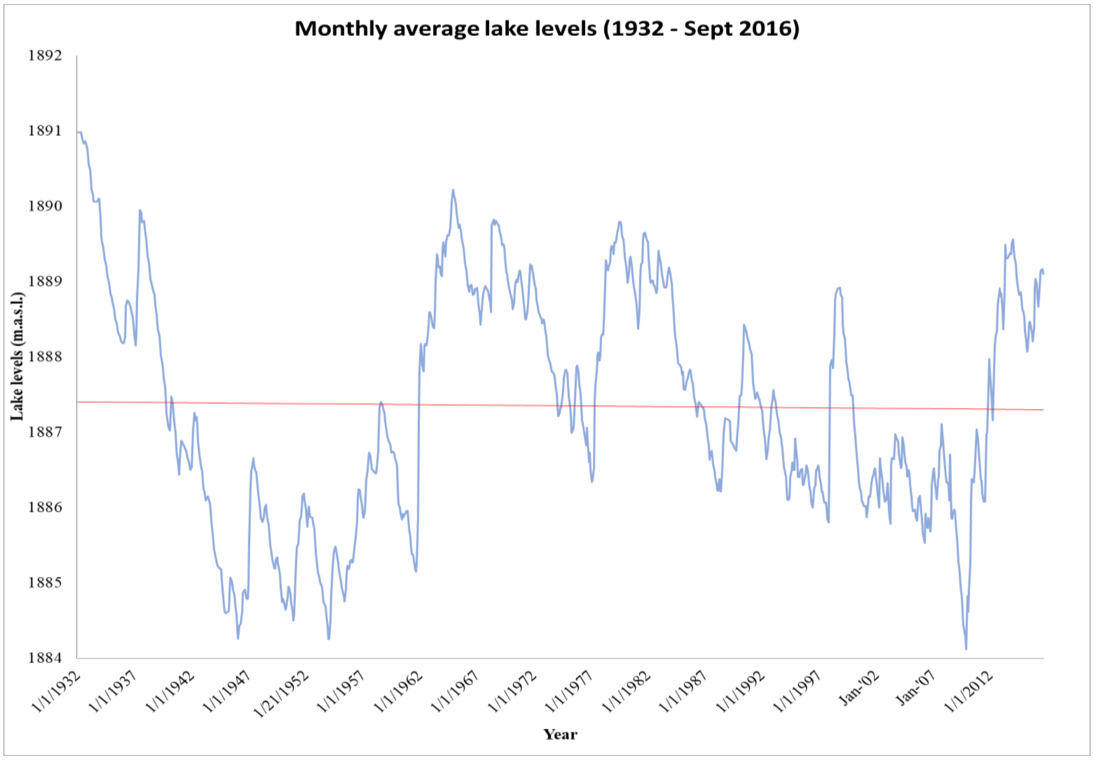
S/N	**Site**	**Core ID**	**Longitude**	**Latitude**	**Core length (cm)**	**Water depth (m)**
1	Malewa	M1	36.339670	-0.728278	129.5	4.2
2		M2	36.339696	-0.752069	135	6.3
3		M3	36.309121	-0.752459	109	6
4	Hippo point	HP	36.318199	-0.788416	144	7.8
5	Middle lake	ML	36.363285	-0.788127	113	6
6	Crescent	CR	36.408070	-0.763718	181	16.3

Appendix II: Procedure on Microwave Digestion

The steps involved for the microwave digestion as modified and summarized from *EPA Method #3051* were;

1. 350 – 500 mg of dried, homogenized sediment samples were weighed and placed in a Teflon digestion vessel
2. A 2.5 ml of concentrated Nitric acid, HNO_3 (trace metal grade) and 7.5 ml concentrated hydrochloric acid, HCl were added to the Teflon vessel.
3. Blanks were prepared using 500 mg trace metal grade, distilled water plus the acids used in step 2.
4. The digestion vessels were capped and placed in the microwave carousel and they were then well secured into the microwave.
5. The microwave was started, and the sediment and acid mixture were raised to a temperature of about 175°C in 5.5 minutes and the temperature was then maintained between 175 – 180 °C for 9.5 minutes. This allowed the pressure in the vessels to peak to approximately 6 atm.
6. The vessels were left to cool to room temperature before they were opened.
7. The contents in the vessels were transferred to a 100 ml graduated cylinder and were diluted to 100 ml mark with distilled water.
8. The dissolved samples were then transferred to polyethylene bottles and were stored awaiting analysis on ICP-OES.

Appendix III: Lake Naivasha Water Levels between years 1932 and 2016



Bathymetric and sediment coring equipment

Appendix IV: Lake Naivasha water depth contours generated from 2016 Bathymetric survey

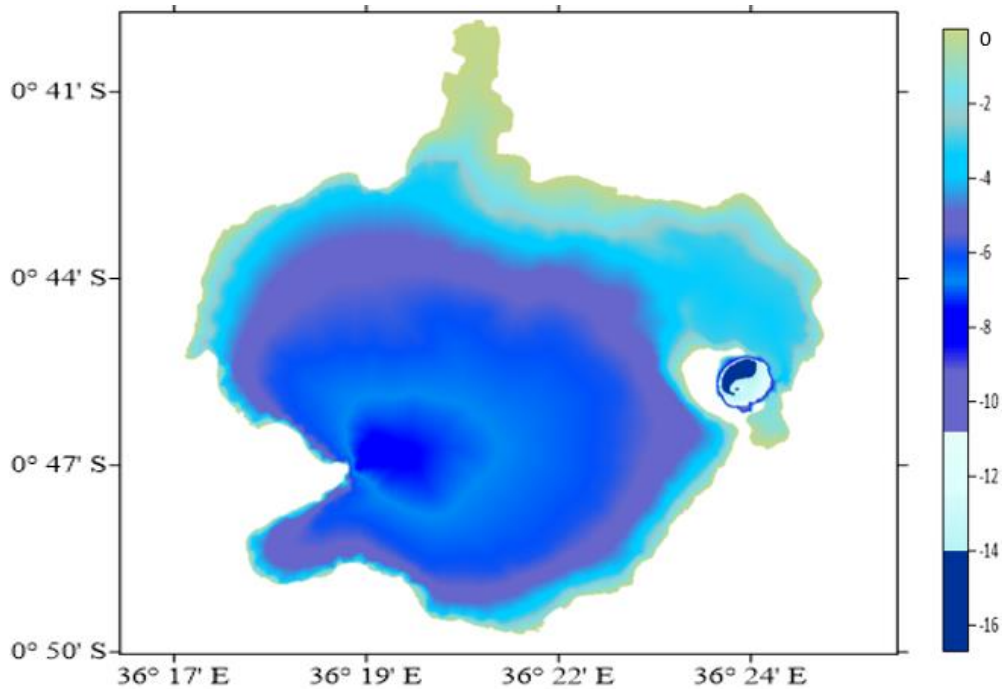


Figure on Lake Naivasha bathymetric water depth map

Appendix V: Characterization of Lake Naivasha basin

Table A5: **Soils within Lake Naivasha basin**

Soil code	Soil_Name	Basin area (%)
PLe	Eutric Planosols	34.78
PHh	Haplic Phaeozems	17.42
ANm	Mollic Andosols	9.18
VRe	Eutric Vertisols	8.82
RGc	Calcaric Regosol	8.12
LVf	Ferric Luvisols	5.94
NTu	Humic Nitisols	4.48
ACh	Haplic Acrisols	4.08
SNk	Calcic Solonetz	3.71
NTr	Rhodic Nitisols	1.55
HSs	Terric Histosols	1.28
LPq	Lithic Leptosols	0.65
SCg	Gleyic Solonchaks	0.01
Total		100.00

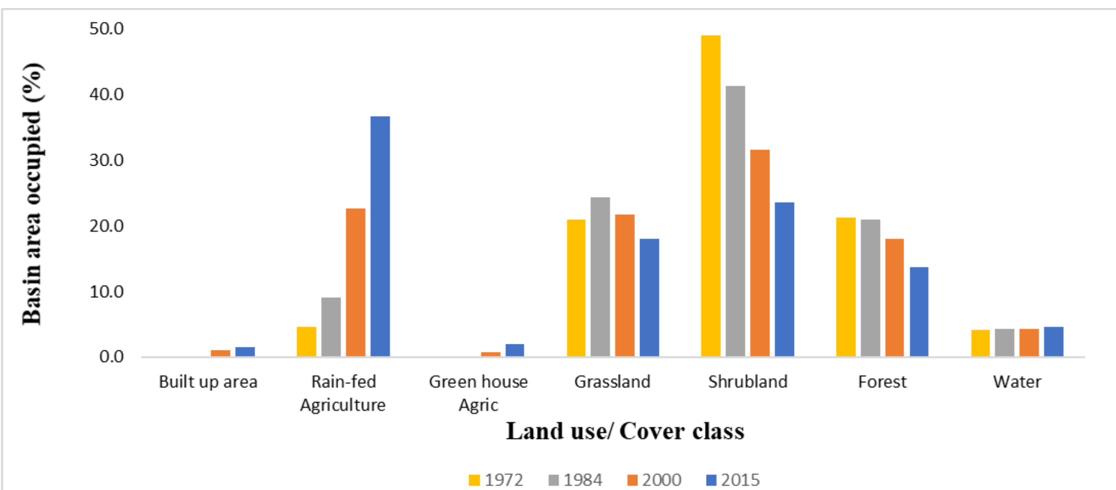


Figure A5: Area occupied by various Land use/ land cover within Lake Naivasha basin between 1972-2015

Appendix VI: Sediment yield per sub basin for the periods between 1981-1992, 1993-004 and 2005-2015

Sub basin	Sediment yield (ton/ha)		
	1981-1992	**1992-2004**	**2005-2015**
1	4.74	5.19	3.76
2	3.24	0.13	2.97
3	0.65	0.85	3.20
4	1.97	6.45	2.16
5	0.05	0.03	0.31
6	1.73	10.39	1.13
7	4.79	14.79	5.64
8	10.08	5.37	13.57
9	0.40	0.93	2.37
10	0.01	0.06	0.04
11	0.41	0.16	0.03
12	0.56	0.12	0.15
13	0.87	2.39	16.45
14	0.17	0.81	1.65
15	0.15	0.00	15.22
16	0.06	0.16	0.06
17	0.21	0.10	0.01
18	0.39	0.03	1.29
19	0.96	2.44	1.81
20	0.02	1.79	0.02
21	0.19	0.75	0.20
22	0.16	0.00	0.53
23	1.48	0.44	1.28
24	2.10	1.28	1.81
25	6.59	1.47	9.27
26	0.03	0.02	0.98
27	0.03	0.09	0.03
28	0.66	0.01	0.48
29	0.04	0.12	0.05
30	0.04	0.02	0.10
31	6.72	3.50	9.52
32	8.75	17.94	22.06

33	0.55	0.02	0.25
34	2.26	5.11	12.08
35	0.86	0.01	0.30
36	1.01	4.46	2.67
37	0.73	0.10	0.52
38	0.29	0.01	0.80
39	0.16	0.00	0.77
40	0.18	0.40	0.21
41	7.80	17.75	12.95
42	0.16	0.00	1.08
43	0.24	0.00	0.67
44	14.95	4.83	47.16
45	6.73	0.05	22.26
46	0.25	0.01	1.33
47	0.01	0.00	0.70
48	0.28	0.00	0.82
49	0.56	0.02	0.67
50	0.01	0.00	0.00
51	2.15	1.18	0.04
52	0.03	0.02	0.00
53	0.61	1.15	0.59
54	0.30	0.03	0.46
55	0.20	0.01	0.11
56	5.97	9.18	16.70
57	0.22	2.90	20.73
58	0.14	0.16	0.07
59	0.00	0.02	2.58
60	0.26	0.02	1.67
61	0.78	1.16	0.45
62	1.37	0.41	1.12
63	0.83	0.07	0.52
64	0.59	0.20	3.69
65	0.48	0.16	1.22
66	0.04	0.00	0.22
67	0.42	0.18	22.45
68	0.00	0.00	0.06
69	1.70	0.31	3.13
70	1.30	0.23	2.37
71	0.00	0.01	0.55

Appendix VII: Papers published from this work

Maina, C. W., Sang, J. K., Raude, J. M., Mutua, B. M. and Moriasi, D. N. (2019). Sediment distribution and accumulation in Lake Naivasha, Kenya over the past 50 years. Lakes & Reservoirs: Research & Management, 24, 162-172. 10.1111/lre.12272

Maina, C., Sang, J., Raude, J. and Mutua, B. (2019). Geochronological and Spatial Distribution of Heavy Metal Contamination in Sediment from Lake Naivasha, Kenya. Journal of Radiation Research and Applied Sciences, 10.1080/16878507.2019.1593718. 10.1080/16878507.2019.1593718

Maina, C. W., Sang, J. K., Mutua, B. M. and Raude, J. M. (2018). Bathymetric survey of Lake Naivasha and its satellite Lake Oloiden in Kenya; using acoustic profiling system. Lakes & Reservoirs: Research & Management, 23, 324-332. 10.1111/lre.12247

Maina, C. W., Sang, J. K., Mutua, B. M. and Raude, J. M. (2018). A review of radiometric analysis on soil erosion and deposition studies in Africa. Geochronometria, 45, 10-19. 10.1515/geochr-2015-0085

Maina, C. W., Sang, J. K., Raude, J. M., Mutua, B. and Karanja, J. T. (2019). Bathymetric survey and sediment core analysis for sustainable water resources management. Exceed-swindon regional experts workshop On Modern and traditional methods of water resource management in Africa. Blue Waters Hotel, Durban, South Africa.