ECONOMICS OF ECOSYSTEM SERVICES AND RESOURCE UTILISATION IN SELECTED WATER CATCHMENT ECOSYSTEMS IN KENYA

EREGAE EKUWOM JUSTUS

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Economics of ecosystem services and resource utilisation in selected water catchment ecosystems in Kenya

Eregae Ekuwom Justus

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DECLARATION

This thesis is my original work and has not been submitted for a degree in any other university.

Signature ………………………………… Date ………………………………

Eregae Ekuwom Justus

This thesis has been submitted for examination with our approval as university supervisors;

Signature ………………………………… Date ………………………………

Dr. Paul Mwangi Njogu, PhD

JKUAT, Kenya

Signature ………………………………… Date ………………………………

Dr. Rebecca Karanja, PhD

KU, Kenya

Signature ………………………………… Date ………………………………

Dr. Moses Kirega Gichua, PhD

JKUAT, Kenya

DEDICATION

I dedicate this thesis to my parents, my mother (Esther Nangok Ekuwom), and my father (John Ekuwom Elinga) who, although uneducated, valued education and supported me in my academic path. They are the drive that has pushed me so hard to achieve this highest level of education. This dedication also goes to my beloved family, my spouse (Naomi Awoton), and my children (Shawn, Randy, and Kate) who have motivated me at different stages of the study and thesis development. Out of curiosity, they asked questions; about who I wanted to be and what it means to pursue a Ph.D.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

LIST OF ABBREVIATIONS AND ACRONYMS

ADB	Asian Development Bank
AGB	Above Ground Biomass
AIC	Akaike Information Criteria
ANOVA	Analysis of Variance
ASAL	Arid and Semi-Arid Land
AU	African Union
AU$	Australian Dollar
BGB	Below Ground Biomass
BT	Benefit Transfer
C	Carbon
CBA	Cost Benefit Analysis
CBD	Convention Biological Diversity
CBO	Community-Based Organisation
CCF	Chief Conservator of Forest
CDAP	County Development Action Plan
CDM	Clean Development Mechanism
CEM	Choice Experiment Method
CGEM	County Government of Elgeyo Marakwet

CO_{2e}	Carbon dioxide equivalent
CVM	Contingent Valuation Method
DeFRA	Department of Environment, Food & Rural Affairs
DRSRS	Department of Remote Sensing and Resource Surveys
DUV	Direct Use Values
EMCA	Environmental Management and Coordination Act
EPA	Environmental Protection Agency
ES	Ecosystem Services
ETS	Electronic Trading System
EU ETS	European Union's Emission Trading System
EU	European Union
FAO	Food Agricultural Organisation
FEVD	Forecast Error Variance Decomposition
FGD	Focus Group Discussion
FPES	Forest Provisioning Ecosystem Services
GDP	Growth Domestic Product
GEF	Global Environmental Facility
GIS	Geographic Information System
GLM	Generalised Linear Model

GNP	Gross National Product
Ha	Hectares
HELB	Higher Education Loans Board
HH	Household
HPM	Hedonic Pricing Method
HQIC	Hannan Quinn Information Criteria
ICAO	International Civil Aviation Organisation
IEET	Institute of Energy and Environmental Technology
IPBES	International Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
IUV	Indirect Use Values
JKUAT	Jomo Kenyatta University of Agriculture and Technology
KALRO	Kenya Agricultural and Livestock Research Organisation
KARI	Kenya Agricultural Research Institute (currently KALRO)
KEFRI	Kenya Forestry Research Institute
KFMP	Kenya Forestry Master plan
KFS	Kenya Forestry Service
KFWG	Kenya Forest Working Group

Kg	Kilograms
KII	Key Informant Interviews
KNBS	Kenya National Bureau of Statistics
KU	Kenyatta University
KWTA	Kenya Water Tower Agency
LPG	Liquified Petroleum Gas
LVBC	Lake Victoria Basin Commission
M	Metres
max	maximum
MEA	Millennium Ecosystem Assessment
MFC	Mau Forest Complex
Mg	Mega grams
min	minimum
mm	Millimetres
MoALF	Ministry of Agriculture Livestock and Fisheries
MoE&F	Ministry of Environment and Forestry
NAAIAP	National Accelerated Agricultural Inputs Access Programme
NACOSTI	National Commission for Science, Technology, and Innovation
NDC	National Determined Contribution

NEMA	National Environmental Management Authority
NPV	Net Present Value
NTFP	Non-timber Forest Product
NTFPs	Non-Timber Forest Products
NUV	Non-Use Values
PES	Payment for Ecosystem Services
PFM	Participatory Forest Management
PPP	Purchasing Power Parity
PRA	Participatory Rural Appraisal
REDD+	Reducing Emissions from Deforestation and Forest Degradation Plus
RES	Regulatory Ecosystem services
s	Seconds
SBIC	Schwarz's Bayesian Information Criteria
SEI	Stockholm Environment Institute
SPSS	Statistical Package for the Social Sciences
SWAT	Soil and Water Assessment Tool
t	ton
TB	Total Biomass
TCM	Travel Cost Method

TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
TLU	Tropical Livestock Unit
UNEP	United Nations Environmental Program
UoN	University of Nairobi
USD	United States Dollar
UV	Use Values
VAR	Vector Autoregressive
VCS	Voluntary Carbon Standard
WHO	World Health Organisation
WRA	Water Resources Authority
WTA	Willingness to Accept
WTP	Willingness to Pay

The study used (USD 1 =KES107)

Kenya PPP to USD (USD 1=47.03)

ABSTRACT

Water catchment ecosystems (WCE) are critical in the provision of goods and services that are essential to societal well-being worldwide. This study assessed the stock and flow of ES drawn from the selected water catchment ecosystem (Elgeyo and Nyambene), aimed at making visible their monetary values to enhance awareness creation and advocate for improved WCE conservation. The socio-cultural aspect of the study targeted a population of 400,000 (Elgeyo), and 700,000 (Nyambene) together with state and non-state actors in the two WCE. Household survey employed stratified and simplified random sampling approach, where it sampled 373 and 402 HH for Elgeyo WCE and Nyambene WCE, respectively. Ecological aspect employed geographical information systems (GIS) and remote sensing, supported by field assessments, laboratory analysis, and literature reviews. Forest biomass mapping employed a stratified-systematic cluster approach with nested concentric design, with 48plots (Elgeyo) and 32plots (Nyambene) sampled. Historical river flow data for three sub-basins (Moiben, Ura and Thangatha) sourced from the Water Resources Authority (WRA) database were used for hydrological dynamic modelling. The economic aspect employed conventional valuation techniques, such as market pricing, stated preferences, and benefit transfers (unit and function) to assign monetary value. Data was collected using a mobile application which was transferred to Microsoft Excel, the Statistical Package for Social Sciences (SPSS) version 24 and STATA for processing and analysis. Descriptive statistics were used to summarise the socio-cultural attributes, forest product extraction, and other quantitative data. The data was subjected to a normality test to check for normal distribution. Both parametric and non-parametric were employed for significant difference and similarity testing. Logistical regression was used to determine the forest's community dependency. Generalised linear model (GLM) was considered in land-based biomass assessment, while VAR was utilised in river flow dynamics assessment models. The study estimates the total ES value at KES 58.8 billion (USD 549.7 million) and KES 39.4 billion (USD 368.4 million) for the Elgeyo and Nyambene WCE, respectively. This translates to KES 542,793.97 (USD 5,072.84) and KES 1,3 million (USD 12,152.99) ha^{-1} $year^{-1}$. Overall, disaggregating the total value on a per capita income, it corresponds between KES 42,416.67 (USD 396.42) and 53,230.77 (USD 497.48) equivalent to 19.4% and 24.4% of Kenya's per capita income. The study estimates indirect use values at KES 90,042.89 (USD 841.52) and KES 48,803.48 (USD 456.11) HH^{-1} $year^{-1}$, respectively. This translates to between 33% and 35% of the forest community's household income thus high forest dependency. Notably, forest dependency is largely influenced by household socioeconomic and cultural attributes. For instance, low-income communities, larger households, and large-scale herders heavily depend on forest resources. These findings imply that the WCE contribute over 30% to rural household income and between 10% and 25% to national gross domestic production (GDP). Equally, the study shows that land cover change impacts on stock and flow of ES as exhibited in assessment of forest biomass and river flow dynamics. For instance, the decrease in forest cover per year results in decline in base flow by between $1mm^3/sec$ and $10mm^3/sec$ while increasing peak flows to between $16mm^3/sec$ and $70mm^3/sec$. Likewise a unit change in forest species diversity, forest cover, and stem volume attributed land cover change would reduce unit forest biomass by a factor of 1.1, 2.2,

and 1.2 on average, respectively. This demonstrate that, a conversion of forest to other land uses, would impact negatively on stock and flow of ecosystem goods and services. Also, the study found out that tree cover change can be a good predictor for river flows changes in sub-basin with minimal anthropogenic pressure. However, it may not be a good predictor basin with facing anthropogenic pressure since its impact becomes insignificant in influencing hydrological flow dynamics. Overall, the outcome of the study, though not absolute, has shown potential monetary values of ES drawn from forested catchment ecosystems in the country. This is thus critical in complementing other conservation efforts that would guide decision-making, since every decision-making process involves trade-offs. Therefore, ES monetary units would be fundamental if a society endeavours to pursue, argue, and justify the need for sustainable water catchment ecosystem conservation in Kenya and beyond. Further research can consider expanding the scope of ES to include economic value on seed dispersal, pest and diseases control among others. To reduce pressure on state forest and enhance stock flow of ES, conservation actors can advocate for farm forestry and enforcement of 10% woodlot establishment policy. Similarly, governments can consider incorporating the economic values of ES into future national accounting and planning processes.

CHAPTER ONE

INTRODUCTION

1.1 Background to the study

Forested ecosystems generate goods and services that are the essential building blocks for societal well-being (Krieger, 2001). Watersheds are part of these forested ecosystems that provide goods and services (Baral et al., 2017; Forslund et al., 2009; Vincent et al., 2016) fundamental to societal welfare (Costanza et al., 1997; Daily, 1997; de Groot et al., 2012; Deal et al., 2012; MA, 2005). These ecosystems support biodiversity (IUCN, 2001; Kumar, 2012), livelihoods, and the economy (de Groot et al., 2012; Deal et al., 2012), especially in poor rural communities bordering forests (Bwalya, 2013; Mamo et al., 2007; Mukul et al., 2016). Like any other asset class, a change in the natural state of the ecosystem will either increase or decrease stocks and the flow of benefits. There is evidence that humans are unsustainably exploiting natural ecosystems, negatively affecting conditions and the flow of benefits. Literature attributes this to, among others, the lack of an explicit assessment of ES, so society cannot make rational informed decisions (DeFRA, 2007). Resource economics seeks to understand the level of resource scarcity and how to allocate finite resources to infinite human demand and needs (TEEB, 2010b). ES assessment not only triggers a public preference for the state of the ecosystem change, but also reveals the societal trade-offs to conserve nature. This will go a long way in reducing the cost associated with ecosystem disruption, which would not differ from the human-made commodity.

Globally, forest ecosystems provide tangible products, including food, water, fuel energy, natural medicines, and building materials (Angelsen et al., 2014; Babulo et al., 2009; Balmford et al., 2002; FAO, 2010; Fikir et al., 2016). And, intangible goods and services, including global climate regulation, water flow regulation, wildlife refugia, and regulation of atmospheric gas chemistry (Daily, 1997a; Deal et al., 2012; Vo et al., 2012), among other benefits. This is, besides being, a source of income and employment for the rural population (Babulo et al., 2009; Bahuguna, 2000; Kabubo-Mariara, 2013; Mamo et al., 2007; Shackleton et al., 2007). For instance, in Asia and the Americas, it accounts for between 10 and 20% of household income (Córdova et al., 2013; Mukul et al., 2016; Uberhuaga et al., 2012).

In sub-Saharan Africa, forested water catchment ecosystems contribute over 40% of household income (Appiah et al., 2009; Kalaba et al., 2013). In Kenya, catchment ecosystems support communities next to forests by providing fresh water, food, grazing land, medicine, timber, and fuelwood, among others (CGIAR, 2015; Shackleton et al., 2002). This is besides contributing approximately 80% of the country's energy hydropower (MoE & F, 2004). Overall, the watershed ecosystems contribute directly and indirectly to approximately 36% of the country's GDP by supporting key economic sectors, including agriculture, manufacturing, trade, and tourism (MoE & F, 2004, 2019a).

Despite the enormous contribution to human well-being (Villamagna et al., 2013), most countries have ignored ES in national development planning (Millennium Ecosystem Assessment (MA), 2005; S. Smith et al., 2013). Even where they have attempted to undertake ES assessments, studies have commonly undervalued them (Costanza et al., 2017). The other challenge is the public nature of ecosystem goods and services attributed to the 'tragedy of the common' (Hardin, 1969) which has resulted in overexploited, with little effort to restore degraded ecosystems. The invisibility and undervaluation of natural resources have made it difficult for a conservationist to support the calls for improved resource appropriation, reject environmentally damaging policies, and promote sustainable conservation (DeFRA, 2007). Environmentalists have identified this as the major setback for the sustainable conservation of forest ecosystems (de Groot et al., 2002; MA, 2005), overexploitation of natural resources, degradation and conversion of forest land to other land uses (de Groot et al., 2002; Eregae et al., 2021). This has led to a decline in the stock and flow of ecosystem services (Millennium Ecosystem Assessment (MA), 2005; Shaw et al., 2011) thus threatening the very livelihood (Barbier, 2015; Mutoko et al., 2015; van Jaarsveld et al., 2005) that depend on them (Sutherland et al., 2018; Villamagna et al., 2013). Similarly, a partial ES assessment would cause an inability to understand natural capital and the trade-offs involved (Keeler et al., 2012), which would subsequently lead to unprecedented environmental consequences (Nahuelhual et al., 2007; Sutherland et al., 2018).

While considerable data is available for direct use on a global scale, little is available for indirect use values (IUVs) (Barbier et al., 2009; Carpenter et al., 2006), locally. Primarily, the assessment of the ecosystem was driven by the need to reverse biodiversity loss (de Groot, 1987). However, this has so far expanded to include ES as a tool for policy and policy intervention, environmental impact and project assessment, spatial planning, and conservation education (Di Franco et al., 2021; GIZ, 2012; Smith et al., 2013). The assessment, therefore, aims at showing the level of ecological diversity and ecosystem state that ensures a sustainable stock and flow of ecosystem services and reduces the risks of degradation (Perrings et al., 2006). The link between ecosystem services and human well-being is a manifestation of the societal benefits derived from natural ecosystems, particularly from use values.

Although studies have exhibited diverse forms of utilitarianism, the attribution of values to the ecosystem is based on the satisfaction of human needs and desires (Costanza & Daly, 1992; Goulder & Kennedy, 1997). ES assessment is based on human preferences and changes that are influenced by marginal changes in the supply of quality and quantity of ecosystem goods and services (Pascual et al., 2010). Although perceived as an anthropocentric view, it is best suited to guiding the decision-making process, especially policy development. Anthropocentric-based assessment is also interested in inner values (DeFRA, 2007). This then requires an explicit assessment of ES, which will go a long way in promoting policy options to improve ecosystem conditions and showing the value of investing in natural resources. Notably, the assessment itself is not a panacea, but a complement to other scientific considerations and interventions to support ecosystems and biodiversity conservation (Costanza et al., 2017; Turner & Daily, 2008).

1.2 Statement of the problem

The Elgeyo water catchment ecosystem forms part of the Lake Victoria and Rift valley drainage basin, while the Nyambene forms part of the Ewaso Nyiro and Tana-Athi drainage basins. They are critical in supply of water and other ecosystem benefit to the population within beyond their boundaries. However, like many forested catchment ecosystems in Kenya, they are increasingly being threatened by anthropogenic pressures through encroachment, illegal logging, and forest land conversion, among others. For instance, between 1990 and 2019, the Elgeyo and Nyambene WCE respectively, lost over 4000ha and 300 ha to other land uses (KWTA, 2020b, 2020c). Communities bordering these ecosystems have continued to over-harvest forest resources, more so because they can only understand the direct benefits they get from the forest. Some of these ecosystems have severely been degraded and can longer provide goods and services to society. It was evident that most of the stream emanating from Elgeyo and Nyambene are dry most part of the year, while what used to be permanent river are now seasonal. These rivers record enhanced peak flows, and to some extent flooding during rainy seasons.

In addition, most of the conventional market have only provided price for a subset of ecosystem services, particularly those tangible tradable, thus excluding the biggest portion of intangible ones. National accounting system also excludes ES assessment and valuation in planning its development agendas and decision-making process. These, among other challenges, have portrayed forested ecosystems as less beneficial compared to other land uses thus encouraging forest land use change. These have subsequently resulted in an irreversible loss of biodiversity and other ecosystem benefits thus threatening the livelihoods particularly the forest bordering community that depend on them for sustenance and growth. This was consistent with what has been reported in literature on the impact of degradation of river flows (Dhyani & Dhyani, 2016; Masiero et al., 2019).

In curbing these threats, state and non-state actors have employed a couple strategies to mitigate forest degradation through programs such as enforcement, awareness creation, restoration of degraded areas, and establishing fences, among others. However, these have bone little in addressing WCE degradation largely because the society cannot understand first the monetary value of benefits drawn from these WCE, and cost of degradation. Similarly, the society cannot link degradation to reduction of stock and flow of ES and human well-being. The assessment attempts to estimate the monetary value of ES drawn WCE, the societal cost of WCE degradation as well as demonstrating the link between the prevailing ecosystem status, stock and flow ES in relation to degradation. The overarching aim of this study is not only making visible the value and cost of degradation but provoking the need for enhanced investment, protection, promote sustainable utilisation, and conservation of water catchment ecosystem in Kenya and beyond.

1.3 Justification and Significance of the Study

Water catchments are part of forested ecosystems that provide goods and services that support livelihoods and well-being, particularly for forest-bordering communities (Alamgir et al., 2016; Baral et al., 2017). They are among some of the highest biodiversity terrestrial ecosystems, which support the economy and sociocultural functions worldwide (Alamgir et al., 2016). This is through the provision of tangible benefits, including water, food, fibre, and fuelwood, among others. This is besides the provision of intangible benefits, including pest and diseases control, flood control, soil erosion control, and regulation of climate, among others (Deal et al., 2012; Jacobs et al., 2016; Millennium Ecosystem Assessment (MA), 2005; Sears et al., 2017; TEEB, 2012; Vo et al., 2012).

Assessment and economic valuation of ecosystem services are critical in establishing the missing data and attempting to correct the imperfect markets that studies have often overlooked. This calls for the development of techniques that fully account for natural capital in a metric unit that would be easily acceptable by multiple actors in society. The adoption of monetary and metric units presents an analysis of ecological systems more transparent and can guide the decision-makers on, among others, the relative merits of different management actions (Mooney et al., 2005; Turner et al., 2003). Notably, many economic decisions involve trade-offs between a range of actionable options. Even the very listing of ecosystem services without capturing the monetary value is likely to play a pivotal role in ensuring the recognition of ecosystem goods and services (Costanza et al., 2017). The visibility of ES worth, whether explicit or otherwise, would better justify the conservation argument than without (Costanza et al., 1997).

This study focuses on a local ecosystem-based assessment because the stock and flow of ES vary across landscapes, forest types, and vegetation types (Alamgir et al., 2016; Baral et al., 2014; Burkhard, Kroll, et al., 2012; van Oudenhoven et al., 2012). They also vary based on ecosystem ecological status (de Groot et al., 2002, 2010; Muller & Burkhard, 2012; Seppelt et al., 2011). In addition, the ES assessment techniques used differ, with most studies extensively using unit transfer, proxies, and secondary data (Seppelt et al., 2011). Using unit values stems in part from studies outside of sub-Saharan Africa and has identified some glaring flaws that may mislead target consumers. This is critical since it attempts to address some gaps identified in previous studies, besides the identification of ES, profiling of beneficiaries, and mapping of ecological changes (Burkhard, Kroll, et al., 2012; Egoh et al., 2011; Naidoo et al., 2008; Nelson et al., 2009). Likewise, establishing shadow prices for unpriced ES, correcting distorted data, and understanding trade-offs between land use (Masiero et al., 2019) on a local scale.

Elgeyo and Nyambene are among the critical water catchment ecosystems in Kenya that host unique biodiversity, pristine landscapes, geology, and a diverse range of resources, among others. The two ecosystems directly support over 400,000 households through the provision of fresh water, food, and medicine, among others. In addition to creating employment and the local economy through the support of the local industries (paper, wood processing plants, tea, and coffee), and small-scale traders. They are also critical for global climate amelioration through the sequestration of greenhouse gases, such as CO_2. The choice of the two ecosystems is principally based on their distinct ecological and conservation status, management regime, and accessibility rights. For instance, the Elgeyo represents an exotic and industrial forest-dominated watershed with a hybrid management regime and open access, while the Nyambene represents a native forest-dominated ecosystem with a state-control management regime and restricted access. The two ecosystems represent two distinct characteristics of most of the water catchment ecosystems in the country. Also represent some of the less expansive watersheds which are equally important in supporting societal well-being worldwide. The study aimed demonstrating their economic value while comparing and contrasting the stock and flow ES between the two distinct water catchment ecosystems in the country.

The study also assessed the impact of land use on stock and flow of ES, and researchers have traditionally employed complex and data-intensive models. These types of models demand a high level of knowledge and expertise to generate, analyse, and interpret. Similarly, the commonly employed tree/ stem-based algorithms have recorded limitations since they largely generalise the impact of land use changes on forest biomass. This is primarily because literature builds them around measurable trees and shrubs, compartments, and dimensions (Henry et al., 2011; Kinyanjui et al., 2014). The study opted to develop a unit area-based biomass estimation algorithm to show how, for instance, changes in forest diversity, tree cover, and stem volume impact forest biomass. Likewise, developing a more simpler but robust model for easier prediction of the impact of land use change on the stock and flow of ES in water catchment ecosystems.

1.4 Hypothesis

H_{0a}: The socio-economic traits in forest communities do not significantly influence forest resource use and dependency

H_{0b}: Forest resources do not have a significant impact on community livelihood and the national economy

H_{0c}: The current level of exploitation of ecosystem resources and land use change does not affect the stock and flow of ecosystem services from watersheds in Kenya

H_{0d}: Tree cover change is not a good predictor of forest biomass and river flow dynamics

1.5 Objectives

1.5.1 Main Objective

To assess the economic value of ecosystem services and the impact of resource utilisation within Elgeyo and Nyambene water catchment ecosystems, Kenya.

1.5.2 Specific Objectives

1. To assess the forest community perception and dependency on ES in Elgeyo and Nyambene Water catchment;
2. To estimate the total economic value of ecosystem services for the Elgeyo and Nyambene water catchment;
3. To assess the impact of land use change on the state and flow of ES using biomass and river flow dynamics;
4. To model the impact of forest cover, change on stock, and flow of ES using forest biomass and river flow regimes.

1.6 Research Questions

1. How do the forest community socio-economic and cultural attributes influence the use, perception, and dependency of ES drawn from water catchment ecosystems?
2. How do water catchment ecosystems contribute to both local and national economy in Kenya?
3. What are the detectable impact of land cover and land use change on stock and flow of ES?
4. Can tree cover change be a predictor of stock and flow of ES in watershed ecosystems?

1.7 Scope of Study

The study focused on two of the selected water catchment ecosystems (Elgeyo and Nyambene) which are part of the critical network of watersheds in Kenya. Elgeyo forms part of Lake Victoria and the Rift valley drainage basin. While the Nyambene forms part of the Ewaso Nyiro and Tana Drainage basin. The data collected includes the aspect of socioeconomic and cultural, ecological, and valuation of ES. The socio-cultural aspect targeted forest-adjacent communities within the 5-kilometre buffer zone, including households, state and non-state actors, conservation groups, and forest product traders. In Elgeyo, it drew the target population from Keiyo South and Keiyo North in Elgeyo Marakwet County, Ainabukoi, and Moiben in Uasin Gishu

County for the case of Elgeyo. In Nyambene, the study drew the sample respondent from Igembe South & Igembe Central, Tigania East, and Tigania Central in Meru County. Socio-cultural data focused on daily household extraction amount extrapolated to annual estimates. This is besides the assessment of the socioeconomic attributes and household contextual factors primarily to determine forest dependency.

Ecological components included field-based plant and soil carbon mapping, river flow dynamics and land cover/use, and change supported by secondary data. The study also generated a simple model of river flow dynamics and forest biomass as a function of land cover change. The economic assessment used market prices for directly used products, cost-based techniques for indirect-use values, and stated preference for non-use values.

1.8 Limitation of Study

Although the study addressed some of the glaring gaps cited in earlier studies, it still records some limitation. For instance, the socioeconomic data, particularly household data, was based on respondents' memory and willingness to give the actual information. Some products, such as timber, fencing poles, and building materials not harvested regularly, so the data provided would depend on the respondent's ability to recall. Illegal extraction which most of the respondents remained silent, and therefore the study may not fully accounted for all extraction of the forest products. Nonetheless, the study captured all the relevant data from all available sources including key informant, informal meetings and complemented with the secondary data.

On the valuation front, only subsets of ES are traded in conventional markets, such as wood forest product, have defined unit prices. However, society does not trade, indirect use values such as water regulation, soil erosion control, and pest and disease control in conventional markets. The lack of market prices combined with insufficient data dictated the use of surrogates, such as water reservoirs as a substitute for ecosystem watershed protection function, dredging cost as a substitute for sedimentation control, and artificial fertilisers as a substitute for soil nutrient conservation among others. Some of these surrogates, while used in some studies, may not apply in all cases, regions, landscapes, and ecosystems. Locally, such data is unavailable and thus the study relied on grey literature, which may be an unclearly defined assessment method (Balvanera et al., 2012). This demanded a back-and-forth literature review in tracing the method's reliability.

Primarily, market pricing would be the most appropriate valuation technique, particularly for ecosystem products traded in markets since it primarily relies on the production and or cost data thus easy to generate (Ellis & Fisher, 1987). However, the lack of local market prices for most of the products and distorted market data posed a change in determining the economic value of some products. Equally, the revealed preference technique, which would be an alternative to market pricing (Kontoleon & Pascual, 2007) at a time being criticised. This is largely because of its demand for large data and more sophisticated statistical analysis to assess and estimate ES value (Pascual et al., 2010). The study, however, mitigated some limitations through rigorous literature reviews before settling on the most appropriate and acceptable surrogate and the technique for valuing indirect and non-use values.

The study also employed function transfer, which again has its caveats, mainly because of the difference in ecosystem attributes, human preferences, and diverse beneficiaries, which vary from landscape to landscape. Inappropriate application of benefit transfer and surrogacy without considering the variability of ecosystems and how they interact with humanity at different times and scales also poses assessment credibility. Even where commodity prices were available, still found inconsistency with the recorded extreme. The application of such prices may have affected the overall ES value. Equally, the pluralism of the assessment method for the reference studies posed a challenge, in terms of comparison of the study findings with the reference literature consistent, as Mengist et al. (2020) emphasised in their study. The study, however, applied average values generated from a wide range of studies to mitigation the likelihood overestimation and underestimation.

There was also a challenge in targeted community groups in understanding the ES concepts, besides the hypothetical markets and complex ecological functions. These scenarios were consistent with what studies have reported in the literature

(Kontoleon & Pascual, 2007; Svedsäter, 2003). Equally, the study is not devoid of the big unanswered question on the principle behind the hypothetical values and objects of choice in real-life situations. For instance, whether values of the non-use services estimated using stated preference are commensurate with the actual monetary value remains unanswered, similar to what other studies have reported (Carson et al., 2001; Martínez-Alier et al., 1998). In this aspect, the study took a lot of time explaining and demonstrating the concept of ecosystem services in simpler way before allowing the respondent to choose from among the bids quoted.

The other challenge was the bundling of ecosystem services and lack of sensitivity to variability in terms of ES assessed by the previous reference studies. This is besides the focus on urban ecosystems, while fewer studies have been undertaken on wetlands, grazing areas, and marine ecosystems. The scenario made it difficult to compare the study with the previous ones. This corresponds to what earlier studies, including Mengist et al. (2020), and others (Diamond & Hausman, 1993; Kahneman, 1986; Kahneman & Knetsch, 1992; Svedsäter, 2000) besides extrapolation of a minor component of ecosystems to represent a larger ecosystem (Boyle et al., 1994; Veisten, 2007).

Although the study reported economic values as total, it worth to note that it did not include some other indirect use values such as seed dispersal, pest and disease control, and refugia, among others. It largely attributed this to, among others, a high demand for large data, complex ecological systems and scanty information on some of these ES thus not able to link benefits the subject ecosystems. In line with this, the study estimates are largely indicative and conservative. That notwithstanding, the study confirms that findings were appropriately estimated using the most suitable and commonly applicable materials and techniques in valuation of ES.

1.9 Theoretical and conceptual framework

An ecosystem is a dynamic complex making biotic components, including microorganisms, plant, and animal communities; and abiotic components that interact as a functional unit (MA, 2003). These ecosystems generate benefits (Costanza et al., 1997; Daily, 1997b; Millennium Ecosystem Assessment (MA),

2005) that are critical to human socio-cultural, economic, and well-being (de Groot et al., 2012; Deal et al., 2012). The benefits include consumptive direct-use products such as food, water, fuel, fibre, and fodder; non-consumptive direct-use values such as spiritual and aesthetic services; and indirect-use services including water regulation, climate regulation, disease, and pest control, soil erosion among others (Haines-Young & Potschin, 2010; MA, 2003). Although societies focus primarily on instrumental ecosystem services, intrinsic values such as existence are also of interest. Humans interact with ecosystems either directly or indirectly, although their level and extent of interaction gradually keep changing. The dynamic change of human interaction is driving ecosystem change and coupled with natural forces, negatively affecting the stock and flow of ES. Literature primarily attributes this to, among others, compromised ecosystem health and functionality.

The science of ES often refer natural ecosystems as "natural capital" primarily borrowing from classical economics, which defines capital as a stock that yields benefit overtime (Costanza & Daly, 1992). The use of the term capital resonates with the human economic perspective and linking with the ecological perspective. For a society to realize the actual benefit of natural ecosystem (natural capital), it has to interact with human influence forms of capitals. These include human capital, built capital, and sociocultural capital (Costanza et al., 2017). The interaction between the four capitals, however, is not straightforward because it involves a wide range of complex process. This implies that the final benefit to society undergoes a couple of interactions between the ecosystems and human influenced capital. In that regard, the stock and flow of ecosystem services and its linkages to human needs and wants can not be represented in a linear cascade, as commonly reported in some literature. The conceptual framework in this case should demonstrate the crucial interaction between the four capitals where (Figure 1.1). Such a representation should exhibit the linkages between services drawn from ecosystems, their interactions and feedbacks from the human influenced capitals to meet human needs and demands. In the context of this study, it will be misleading to simply denote and categorized a variable as dependent or independent considering there are back and forth complex interactions. However, this will be possible if narrow down to the specific ES.

That notwithstanding, natural capital is crucial to human wellbeing, which constitute basic needs, freedom to choose, good health, social relation, and security (MA, 2003). The status of ecosystems' health and functionality is principally associated with societal preferences, freedoms, and choices available to people and vice versa. This is driven by a couple of factors that vary across different human interests, thus making it both dynamic and subjective. Worth to note, human needs and wants, existing institutional laws and governance structure, perceptions, restoration plans, investments and economic production, it dictates the natural capital status. Negative interaction results in overexploitation pollution, forest degradation, land use and land cover change, among others, and *vice versa*. These factors, besides natural phenomenon, will influence ecological process and functionality, flow of benefits thus impact on human wellbeing (Millennium Ecosystem Assessment (MA), 2005; TEEB, 2010a). Overall, any interference of natural capital process and functionality would mean the destruction of humanity.

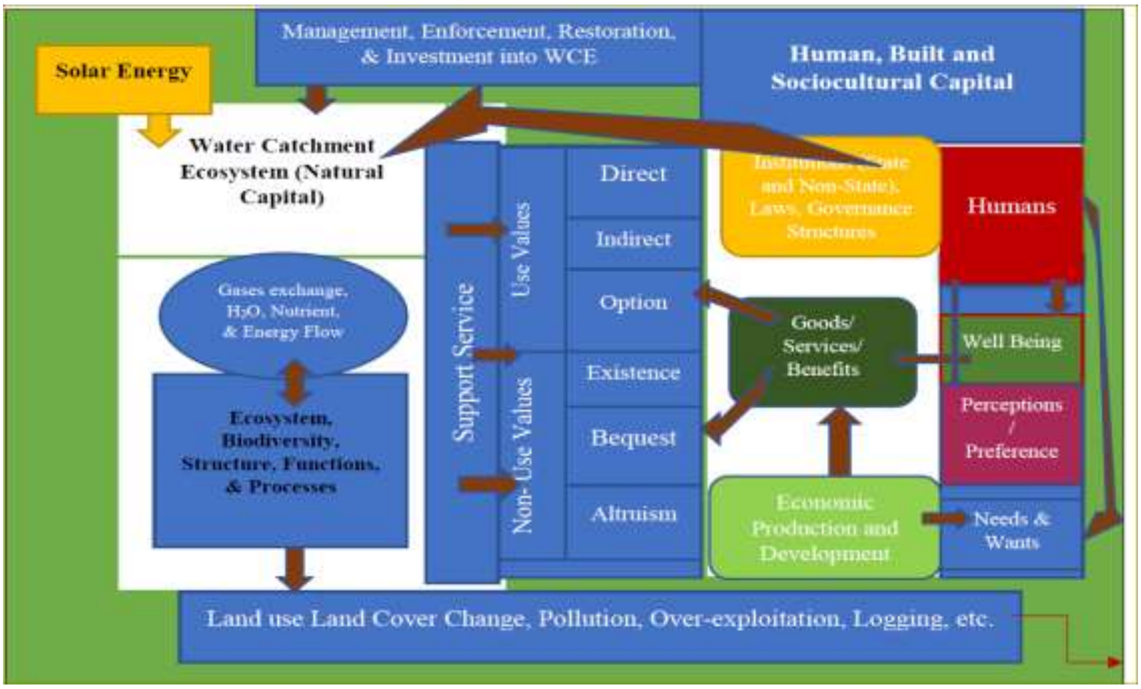

Figure 1.1: A concept on the flow of ES as depicted by a dynamic system exhibiting complex interactions driven by the flows of energy, matter, and human action. (Adopted from Costanza et al. (2017))

The continued loss of forest ecosystems has caused the need to understand their contribution to human well-being and welfare. Assignment of monetary value to ES is an essential and important because it is the most comparable unit of measure, particularly in the assessment of trade-offs. This will also show the contribution of ecosystems to livelihoods and the distribution of costs and benefits to communities. In addition, assignment of monetary value would succour in evaluating policy intervention, their impact and design strategies for demonstrating benefits (Morse-Jones et al., 2011). The application of value to goods and services varies across philosophical disciplines where, sometimes, it could be vague and complex to compute. However, in classical economics, the assignment of value to goods or services is based on consumer preference. Consumer preference is a set of assumptions that focus on consumer choices that can lead to the attainment of different alternatives, such as utility or happiness. This would allow humans to rank or score different goods and services based on the relative score of satisfaction. Notably, the human level of resource use may change based on behaviour associated with societal, technological, and market advancement. The societal choice and decision would either promote overexploitation or sustainable resource conservation (Stainback et al., 2011) and so does the stock and flow of ecosystem services. Thus, this study becomes necessary to demonstrate monetary value for ES and the cost of degradation.

CHAPTER TWO

LITERATURE REVIEW

The chapter discusses the economic principles and theoretical concepts that underpin valuation studies, besides previous relevant studies. The chapter disaggregate this into theoretical principles and previous relevant studies. They include an overview of ecosystem services, ecosystem threats, classification, significance for valuation, valuation techniques, review of ecosystem services typology, and forest dependency, as presented below.

2.1 Theoretical principles

This section presents important aspects and principles of ecosystem services concepts, including the history, assessment, and valuation of ES. This is besides understanding the dictates of values, the potential of double counting, the principle of marginality, interconnectedness, and non-linearity of ecosystem services.

2.1.1 Ecosystem Services History and Concept

Interest in ecosystem services has grown over time, though the most noticeable in the 21st century is the Millennium Ecosystem Assessment (MA, 2005). The ecosystem services have a long history than earlier reported (Grumbine, 1998). In 1970, the Study of Critical Environmental Problems (SCEP) introduced the concept of ecosystem services as environmental services. The SCEP study findings were refined by Holdren and Ehrlich (1974) referred to as 'public service functions of the global environment' (Holdren & Ehrlich, 1974). Later changed to 'nature's services' (Westman, 1977) and finally ecosystem services in the '80s (Mooney & Ehrlich, 1997). The Millennium Ecosystem Assessment (MA), started by the United Nations (UN) is, however, reported as the most comprehensive ecosystem assessment. Primarily aimed at establishing the global ecosystem status, services, trends, changes, and their implication for humanity. This is besides proposing a scientific based intervention to enhance the conservation and sustainable exploitation of ecosystems as well as establish their contribution to human well-being. The key finding of the millennium ecosystem assessment was that approximately 60% of the global ecosystems are degraded and exploited in an unsustainable manner. Though varying across communities (Díaz et al., 2006), the degradation of nature has a significant impact on societal socioeconomic and cultural development (Millennium Ecosystem Assessment (MA), 2005). In that regard, it found explicit accounting of ecosystem goods and services as necessary to reveal the link between ecosystem services and societal well-being (Secretariat of the Convention for Biological Diversity, 2004). This attempts to answer the question of the type of ecosystem services, their stock and flow, and what threatens their availability. The research community gears toward considering ES assessment in conservation and development decision-making (Haines-Young & Potschin, 2010).

Primarily, the establishment of ecological economics was with a view of linking ecology, resource economics, and related science, espoused with theoretical academics and indigenous knowledge (Braat & de Groot, 2012a; de Groot, 1987). However, different schools of thought have grown where some view the valuation of ES as a reflection of the willingness of a society to trade off to conserve natural resources (TEEB, 2010b). Others view the ecosystem as natural capital and the benefit generated from the ecosystem as a dividend derived by humans from the natural capital (Costanza & Daly, 1992). The current focus of assigning monetary units and incentivisation of ecosystem services have so far elicited political debate. Some argue studies aim ecosystem valuation at pricing or privatising nature (Costanza et al., 2017). The same has also generated the neoclassical economics paradigm and the market logic believed to tackle environmental issues. Importantly though, society has to make prudent choices to maintain a certain threshold of natural capital and biodiversity to ensure a continued flow of ecosystem services that support functionality and human well-being analogous to choices made on a certain business portfolio to manage risks in returns (Perrings et al., 2006).

2.1.2 Economics of Ecosystem Services

Economics of ecosystem services like classical economics rely on an assignment of monetary units to reveal its scarcity. It exhibits the opportunity cost, benefits flow, and tradeoffs (Pascual et al., 2010). Society is largely in a consensus that natural ecosystems are 'valuable' and that should consider their worthiness in decision-making processes, irrespective of value interpretation (Daily, 1997b). Scholars have primarily thought of the concept of economics of ES from an anthropocentric and utilitarian perspective, that is more focus on instrumental benefits over the intrinsic value of nature (Mc Cauley, 2004). This is based on the extent to which a society appreciates a product (Costanza et al., 2017; Masiero et al., 2019), and the utility drawn from them. Equally, ecosystems are part of the natural capital and society views them as goods and services accrued from the capital as dividends. (Costanza & Daly, 1992).

However, environmental philosophy and ethics appreciate three sets of values, including instrumental verse intrinsic, anthropocentric verse biocentric (eco-centrism), and utilitarian versus deontological (Callicott, 2004; Gagnon Thompson & Barton, 1994; Oelschlaeger, 1997). The instrumental values represent the usefulness derived from ecosystem services, such as fuelwood, among others. While the intrinsic values reflect the worthiness of the existence of something irrespective of benefit to humanity, such as biodiversity, habitat, among others; The anthropocentric implies human interest matter while biocentric represents ecological system interest; and utilitarian implies the ability to provide welfare underpinned by human preference independent of the relative output; while deontological implies the right to exist (Heal et al., 2005; Masiero et al., 2019). Society has, however, divergent philosophical views on the valuation of nature weighing intrinsic and instrumental ecosystem values. Despite the divergent opinion, studies primarily aimed ES valuation at demonstrating how humans benefit from and how their actions impact nature (Barbier et al., 2009). Society appreciates the value of ecosystems and pursues explicit quantification of ecosystem services to persuade policy and decision-making processes (Masiero et al., 2019) for sustainability.

In a market-based economy, money is the universally accepted unit of measure, and thus the amount of money that a person will pay for a product equates to the other goods and services that they will trade-off to get the product. Though market price does not reflect the true value of ecosystem goods and services, their respective price is driven by the law of demand and supply. For instance, if the price of an ecosystem good or service increases, the demand for that good or service decreases and vice versa. However, there are cases where the market doesn't capture the value of these goods and services, this causes the adoption of "shadow pricing" in assigning its value (Dasgupta, 2010; DeFRA, 2007; Morse-Jones et al., 2009; Polasky & Segerson, 2009).

Notably, ES assessment is not devoid of challenges which range from the public nature of ecosystem resources, lack of developed markets, and distorted data, among others. Although ecological properties, the socio-political, cultural, and economic context underpins the flow of ES, the jurisdiction of the economic agent influences the monetary value paid (Barbier et al., 2009; Pearce, 1993; Wallace, 2007). The direct use values have well-developed markets with reliable data (Barbier et al., 2009; Carpenter et al., 2006). The contrary is the case with indirect use and non-use values since most of these services are not translatable in economic terms (Morse-Jones et al., 2009). The non-use and ecological values support the functioning of ecological systems. This typology of ES may not benefit humans directly but is critical in supporting the provisioning of other ecosystem services, such as clean water. They are non-anthropocentric and are way above monetary values since they are the 'glue' or insurance values (Costanza et al., 1997). Any assignment of the monetary unit would only estimate apportion which is not equivalent to the total system value (Morse-Jones et al., 2009). Though the valuation of an ecosystem would be critically important, assessment should undertake precautionary consideration in evaluating trade-offs while appreciating the partial ecosystem 'glue' or 'insurance' values.

That notwithstanding, the ES assessment employs different approaches and techniques to quantify and assign a monetary value, based on quantitative or qualitative metrics. The qualitative metrics focus on non-numerical, while the quantitative ones make up numerical metrics and monetary units (TEEB, 2009). Valuation techniques can either be biophysical or preference-based (Gómez-Baggethun et al., 2010; Pascual & Muradian, 2010) (Figure 2.1). The biophysical aspect of ES draws its value from the cost of production and preservation, such as the cost of preserving nature. While preference-based relies on human perception built around human behaviour, individual preference is thus subjective, context-based state-dependent (Goulder & Kennedy, 1997; Nunes & Van den Bergh, 2001). The two approaches, though complementary, provide values for different aspects of ecosystem values. Biophysical takes care of the insurance value (Farber et al., 2002) also referred to as ecosystem resilience value (Fisher et al., 2008; Gren et al., 1994). While the preference-based approach underscores the anthropocentric aspect built around utilitarian principles.

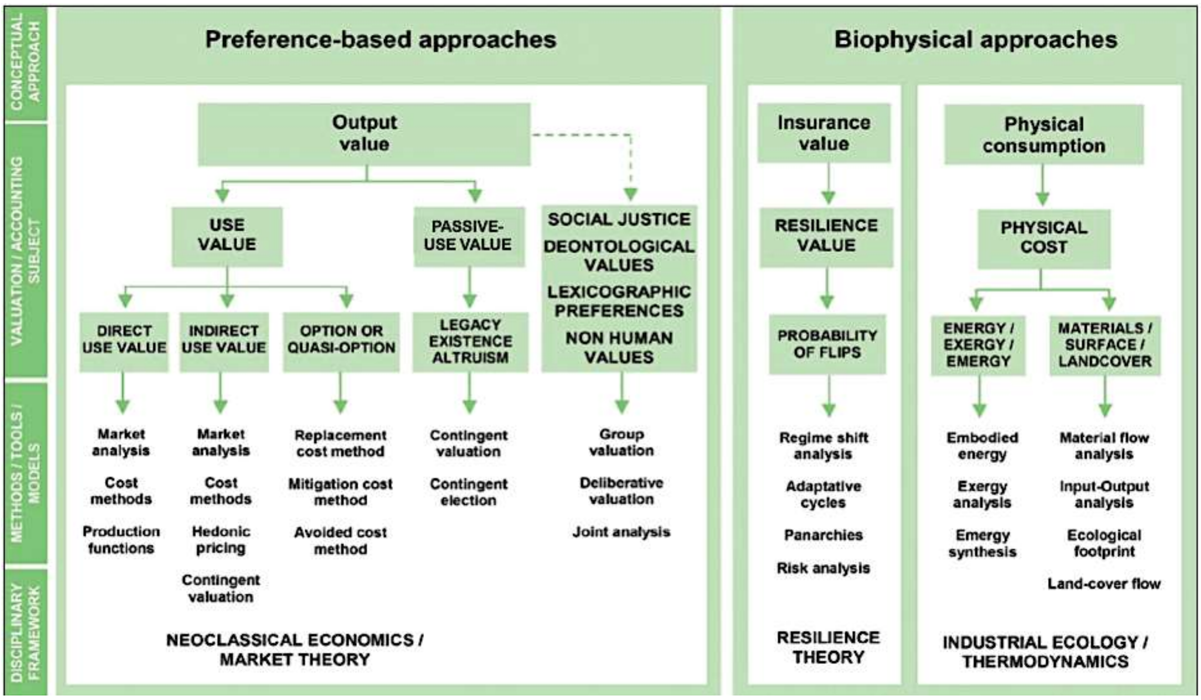

Figure 2.1: Ecosystem Services Assessment Approaches

(Source: TEEB Report (TEEB, 2012))

That notwithstanding, the biophysical approach records limitations related to uncertainties associated with its complexity in assessing and quantifying the related ES (Pascual & Muradian, 2010). This is chiefly the fundamental challenge that explains why more focus is on preference-based approaches, such as market pricing, cost-based, and participatory-based techniques. However, studies have commonly employed utilitarian and non-utilitarian preference-based approaches for the valuation of use and non-use services, respectively. This is despite their utility and contribution to human welfare and both are used to support decision-making despite the difference in metrics employed (DeFRA, 2007; Masiero et al., 2019; TEEB, 2012). The choice of utilitarian approach is primarily determined by ES typology, the time, available data, and resources available. While ethical, sociocultural, and philosophical perspectives dictate the choice of non-utilitarian. The non-utilitarian techniques include contingency valuation, choice model, and participatory commonly employed in valuing intrinsic values and can complement utilitarian techniques. In that regard, any societal consideration and decision made would involve both utilitarian and non-utilitarian values, which make up the concept of total economic value (TEV).

Albeit significant progress in ES assessment, there are still glaring issues, particularly on non-marketed and intangible ecosystem services. Sometimes, researchers apply inappropriate surrogates and this can lead to either underestimating or overestimating. The application of such value is likely to mislead the audience, particularly if used in policy appraisal and policy interventions. In that regard, ecosystem services assessment principles need to be considered and understood to conduct valuation studies. Some of these principles include the what to value, spatial explicitness, marginal and thresholds, the double-counting trap, and nonlinearities in benefits and threshold effects as presented here under.

2.1.3 Spatial explicitness

The stock, flow, and cost of ecosystem services depend on landscape and ecological attributes brought about by geographical ecosystem variability. This, therefore, calls for explicit spatial context analysis in terms of socio-economic, cultural, political, and ecological parameters besides biophysical structure and processes. This will exhibit the both disaggregated and aggregated distribution of ecosystem services across the adjacent areas and beneficiaries. The spatial analysis will also reveal factors that influence the distribution of ES, such as political, distance decay, and environmental traits, among others. According to Naidoo and Ricketts (2010), there is a significant disparity in benefits flow across landscapes brought about by variability in topographical attributes, land tenure, and soil types, among others. The spatial variability influences the willingness to pay (WTP) where, for instance, distance from the ecosystem inversely affects per capita WTP value (Luisetti et al., 2011). This illustrates the criticality of incorporating spatial context in cost-benefit analysis (CBA) to reduce unbiased estimates and determine the distribution of stock and flow of ecosystem services across unique landscapes and localities. Otherwise, if society assumes benefits are similar across the population would be misleading particularly when determining and aggregating ecosystem benefits. The spatial analysis also is critical in designing a choice experiment model used to elicit the willingness to pay, since most of the values quoted are a function of distance bands. Researchers consider this as a guiding principle, principally for ecological planners and managers, to identify new conservation areas and programs (Morse-Jones et al., 2009). Failure to undertake spatial analysis on ecosystem assessment would risk either overestimating or underestimating the economic value of the respective ecosystem. Scholars commonly employ geographical information systems (GIS) to demonstrate the spatial ecological distribution, trends, and ecological characteristics across land cover types or agroecological zones.

2.1.4 Marginal and Threshold Effect

Valuation of ES, like classical economics, focuses on a slight change of stock and flow of benefits as opposed to a big change (Bockstael et al., 2000; Pearce, 1998; Pearce & Turner, 1990; Turner et al., 1998, 2003). Borrowing from the conventional demand and supply model, where the vertices depict the cost or the price, while the x-axis illustrates the stock and flow of ecosystem services (Figure 2.2). The demand curve slopes downward because of the common law of demands that includes substitution effect, income effect, and diminishing marginal utility. These laws dictate how consumers of ecosystem services would pay for an enhanced flow of products. The supply curve slopes upward because of the law of increasing marginal cost, That is as one produces more products, they are likely to use lower quality or more expensive resources. The shape of the graph suggests that as the ecosystem products become scarce, more resources would be required to produce additional units while the contrary is true. Further, as the supply of ecosystem services approaches zero, the demand curve approaches infinity. Ecosystem functions and processes could bring this about by having exceeded threshold levels and cannot support societal demands. Though the study cannot define the area under the demand and supply curve, it represents the aggregated benefit and cost referred to as total benefit and cost, respectively. Similarly, the area between the demand curve and the P1 represents the consumer surplus while the area between P1 and the supply curve represents the producer (ecosystem services) surplus. Commonly economic value for goods and services generated without cost equated to the value of consumer surplus. The stock and flow of ecosystem services are not driven by economic systems (Costanza, et al., 1997).

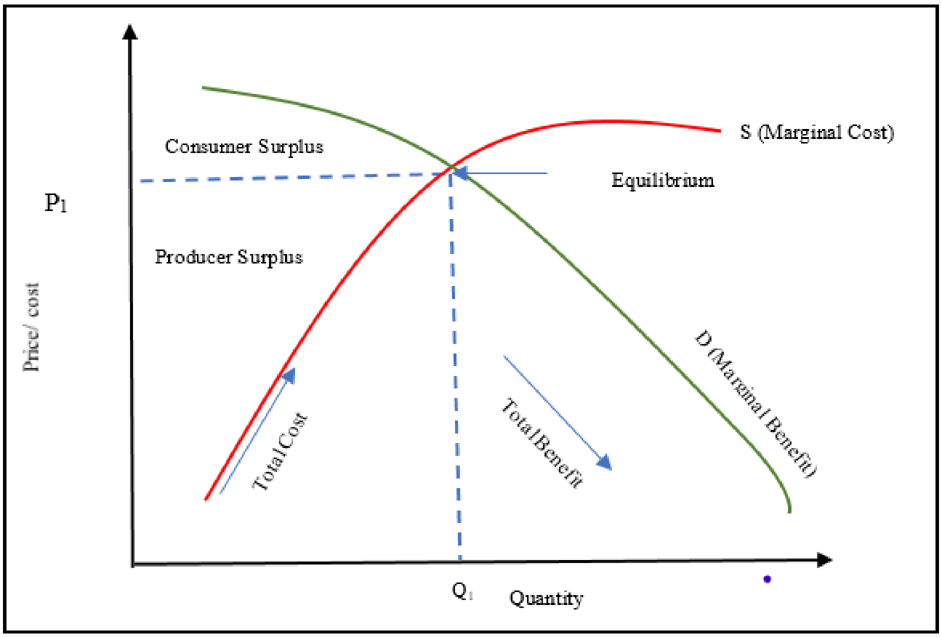

Figure 2.2: Demand and Supply of Ecosystem Services

Source: (Grafton et al., 2004; Morse-Jones et al., 2009)

Valuation of non-marketed goods and services, such as non-use values, primarily relies on estimates of consumer surplus using stated preference valuation methods and approaches (Grafton et al., 2004; Langat, 2016). Establishing marginal changes is complex because of uncertainties associated with ecosystems' functioning and processes, particularly changes beyond the ecological threshold (Turner et al., 1998). Though the 'point estimates' are also important, marginal analysis is currently scanty (Balmford et al., 2002; Bulte & van Kooten, 2000; Turner et al., 2003). This would be more critical in the assessment of trade-offs, the marginal impact of human-induced ecological transitions, and policy intervention (Morse-Jones et al., 2009). However, the focus has been on the valuation of ecosystem stock, such as the economic value of non-timber forest products (Godoy et al., 2000; Peters et al., 1989) and total economic values (Adger et al., 1995; Langat, 2016; Langat et al., 2020; Yaron, 2001). This is contrary to a few marginal change analysis studies, including the marginal value of wetland service in Oregon (Mahan et al., 2000) and mangrove habitat services (Maler et al., 2008).

That notwithstanding, studies consider the marginal analysis, also referred to as 'the next unit' value, with spatial explicitness and policy scale (Fisher et al., 2008). The marginal change would be important if the less ecosystem loss, while the contrary would be true on entire ecosystem loss thus disastrous to humanity. In that regard, a marginal analysis should always consider the spatial extent of policy and policy intervention being undertaken. That is the assessment should delineate policy boundaries either on the local, regional, or global scale (Fisher et al., 2008). This should also inform the valuation techniques employed in the various scenarios (Bateman et al., 2002; Bockstael et al., 2000; Pagiola et al., 2004; Soderqvist & Soutukorva, 2006).

However, when the value of change is beyond the safe minimum standard (SMS) or functional threshold (Fisher et al., 2008b; Turner et al., 2003) it renders the marginal analysis meaningless (Scheffer et al., 1993, 2001). The functional threshold level is a state where an ecosystem can supply services sustainably beyond which the supply of services would approach infinity that society may not accurately account for. In that regard, ecological economics researchers should endeavour to understand the 'tipping point' level, where necessary to undertake the assessment far away from the sharp shift or infinity point, as exhibited by a conceptual supply curve. Ideally, the establishment of an ecological threshold point with its ecosystem functionality complexity notwithstanding (Turner et al., 2003). There will therefore be a need to develop systems and models that can show ecosystem SMS, which goes a long way in facilitating sustainable conservation and management of particularly fragile ecosystems (Lenton et al., 2008). For example, the 'degenerate fingerprinting' model, which is still in the early stages of trials and uses time series output, in testing the shifting of Atlantic Thermocline Circulation (ATC) toward another state (Held & Kleinen, 2004); Modelling of optimum eutrophication in shallow lake ecosystem (Hein, 2006). Data on the flow of ecosystem services, costs of eutrophication control, and lake response to intervention. This is besides the ecosystem economic model that was used to compute the economic thresh values in De Wieden lakes in the Netherlands (Hein, 2006). These are some examples of threshold testing models developed to assess and report on ecosystem status and deter ecosystems from approaching a tipping point.

Notwithstanding the inadequacy in terms of knowledge of ecological complexities and inter-linkages, researchers can use the opportunity cost and analysis models. The conservation option provided, coupled with ethical/ political intent, society could choose and agree on the threshold points. Threshold levels are, in most literature, only acknowledged but rarely included in valuation studies (Dasgupta & Maler, 2003). Importantly, though, society has to make prudent choices to maintain a certain threshold of natural capital and biodiversity to ensure a continued flow of ecosystem services that support functionality and human well-being. This corresponds to choices made on certain business portfolios to manage risks (Perrings et al., 2006).

2.1.5 Double counting

The ecological complexity and uncertainties associated with the connectedness of the ecosystem services are suspect in the double counting on ES assessments (Fisher et al., 2008b). Ecosystem valuations attempt to quantify and value ES either separately, or as an aggregate. The latter, for instance, provides estimates without distinguishing between the intermediate and ultimate benefits. In most instances, ecosystem services studies have aggregated values without considering overlaps and inter-linkages between services and the ultimate benefit (de Groot et al., 2002; Fisher et al., 2008b; Turner et al., 2003). This has the potential to double counting instigating criticism, particularly when researchers aggregate values without eliminating the portion of intermediary services. Few studies have attempted to avoid double counting by distinguishing and separating intermediary services and the ultimate benefits, such as the cost benefit analysis (CBA) of the United Kingdom (UK) coastal managed realignment policy (Turner et al., 2007); other benefits provided by salt marsh (Luisetti et al., 2008) and meta-analysis (Woodward, RT & Wui, 2001). In that regard, researchers need to first appreciate the inter-linkages and overlap before aggregating the ecosystem's benefit values (de Groot et al., 2002; Turner et al., 2003). Only incorporate regulatory ecosystem services in aggregate values, where reported the impact of the ecosystem change goes beyond the ecological boundary and benefits the adjacent community (Hein et al., 2006). Overall, to avoid double counting, valuation studies should adopt a classification system that clearly distinguishes services and that the ultimate benefit should be the only component subject to valuation (Fisher et al., 2008b; Morse-Jones et al., 2009) and incorporated in aggregated values.

2.1.6 Ecological Interconnectedness and Nonlinearities

It is necessary to enhance and maintain links and genetic trade-in, because of ecological interconnectedness, especially between populations when undertaking ES assessments. The connectedness underscores the functionality of an ecosystem, either directly or indirectly. For instance, the degradation of a watershed ecosystem has a spiral effect on the availability of water downstream. This is primarily because of interference in a short hydrological cycle or increased flooding and sedimentation. Equally, flooding pulse will, as a result, create spiral nutrients downstream that are likely to affect the fish population from the organic matter and nutrients deposited (Junk et al., 1989). The stability and resilience of an ecosystem depend on diverse interactions of organisms at any level. Literature structures the uniqueness of how communities based on multiple biotic processes and the condition thus influence the ecosystem output (Griffin et al., 2009).

Diverse ecosystems respond to perturbation differently because of their unique ecological attributes. Unique ecosystems respond to stressors differently and, depending on the extent of change, the services generated by such ecosystems change non-linearly (Barbier et al., 2008). Equally, the non-linearity and complex ecological processes and functions make ecosystem valuation more challenging, particularly in tracing the impact of smaller changes on the ecosystem. In that regard, ecosystem changes may go unnoticed until a drastic change occur that may lead to a complete shift in status (Arrow et al., 2000; Turner et al., 2003), and the impact becomes more apparent. For instance, small-scale harvesting of timber in forested water catchment ecosystems would be very hard to linearly link to slight changes in river flow and water discharge. This may go unseen until there is a drastic change that leads to a change in indicator services, such as river flow and discharges that researchers can easily quantify and link to the ecosystem change. In that regard, one cannot assume that the provision of goods and services, individual services, is uniform across landscapes. For instance, society cannot assume enhanced fish provision on unit habitat improvement to be constant because of the nonlinearity of the response of different aquatic ecosystems to habitat improvement (Barbier et al., 2008). The non-linearities in ecosystem services flow to provide an opportunity for a wide range of policy options for society to choose from. Studies make choices through cost-benefit analysis and on which the policy option makes economic sense regardless of the non-linear outcome or otherwise.

Overall, the ecosystem generates a range of interlinked goods and services and thus does not encourage the generalisation of change on stock and flow, because of the non-linearities in the ecosystem's response to change. But when undertaking an explicit spatial analysis with a consideration of ecological complexities and limitations. The expected ecosystem services flow outcome depends on a wide range of factors, including the extent of the ecosystem change, geographical location, and the prevailing status of the ecosystem (Morse-Jones et al., 2009). Overall, despite the uncertainties in the analysis, valuation studies should endeavour to generate marginal values to establish and link ES outcome with the relative to ecological change non-linearity notwithstanding.

2.2 Previous works relevant to study

The section presents the literature review from relevant studies undertaken around the globe, touching on, among others, the ES concepts and assessment, valuation, significance of ES studies, techniques, classification typology, and valuation of direct use and indirect use values.

2.2.1 The Ecosystem services concept, valuation, and existing gaps

The concept of ecosystem services is becoming popular in empirical science, nature conservation, and policy discourse (Braat & de Groot, 2012b; Fisher et al., 2009a; Seppelt et al., 2011). Such studies aim to understand the contribution of nature to households and community livelihoods, the economy (Boyd & Banzhaf, 2007), and the maintenance of life on earth (Deal et al., 2012) with the ultimate intent of sustainable conservation. Different authors have diverse opinions on the definition of ecosystem services (Costanza et al., 2017). According to Mooney and Ehrlich (1997), ecosystem services are the intertwined conditions and functions that an ecosystem fulfils and sustains humanity. While other scholars define ES as benefits that humans gain from nature (Costanza et al., 1997; de Groot et al., 2002; Millennium Ecosystem Assessment (MA), 2005). Boyd and Banzhaf (2007) define it as ecological components that support human well-being. Overall, ecosystem services are associated with ecological characteristics, functions, or processes of a functioning ecosystem that directly or indirectly contribute to human physical, socioeconomic, and cultural well-being (Braat, 2013; Costanza et al., 1997; Daily, 1997a; Millennium Ecosystem Assessment (MA), 2005; Palmer et al., 2004; TEEB, 2010a). Notably, though, the ecosystem functions and processes are distinctively different from ecosystem services and disservices. For instance, the ecosystem functions and processes represent and define the ecological system regardless of human benefits or otherwise. While the ESs are the output of the ecological process and functions that benefit man either directly or indirectly (Braat, 2013; Costanza et al., 2017). Whereas, the disservices denote outputs that cause harm and incur costs to humans, for example, an outbreak of plagues, pests and diseases, and floods, among others (Sandbrook & Burgess, 2015; Shapiro & Báldi, 2014). However, the linkages between ecological processes and functions with ecosystem services are not straightforward and clearly not well understood (Costanza et al., 2017), though researchers have made notable progress. Likewise, the relationship between ecosystem functions, processes, and associated benefits remain one of the glaring questions in ecological research (ICSU, UNESCO, UNU, 2008).

According to Miriam Webster (2022), value is the monetary worth of something for those with a market price or a fair return or equivalent in goods and services, or money. Valuation, therefore, is a quantification of the worth, utility, or importance of something essential in the perception process. This involves analysing, comparing, and deciding on situations and a reference value when deciding consciously or unconsciously based on societal needs and desires (Farley, 2012). The assignment of monetary value to goods and services is driven by human desire and needs, either as individuals or in groups. It can also express this through appreciation of nature, such as visitors in a game park would pay a gate fee as a sign of appreciation for biodiversity. Society should not construe such an appreciation to make a trade-off (Costanza et al., 2017) but the basis of demonstrating the somewhat worth of intangible benefits (Braat et al., 2014). Economic valuation is based on how ecosystem services contribute to human welfare and estimate it as aggregates of individuals' assessment of well-being (Bockstael et al., 2000; M. Freeman, 2003).

Valuation of ecosystem goods and services is principally based on a monetary unit, an individual place on the respective commodity to acquire or prevent the loss of goods and services based on their preference. In that regard, the worth of ES is the maximum or minimum amount of money an individual would pay and accept as compensation to cease a benefit or lose it (Langat, 2016). The Millennium ecosystem assessment (2003) expressed this in three value domains, that is sociocultural, ecological, and economic. Socio-cultural values include societal culture, social identity, and spiritual attachment to nature. While the ecological values include a function of the natural system assessed using ecological indicators, such as the state of biodiversity and vegetation cover, among others. Whereas the economic aspect reveals a monetary unit of the product.

Currently, conventional markets only transact and provide prices for a subset of ES, particularly those that are transacted in local markets (Gardner & Prugh, 2008). This excludes the larger portion of ES the indirect use values (Bishop, 1999), thus failing to account for ecological systems in entirety (Millennium Ecosystem Assessment (MA), 2005). This poses a structural limitation since the markets cannot show an explicit ecological value, thus not able to fully inform the decision-making (Pascual et al., 2010). Equally, the ecological assessment data is scarce, and even what is available has gaps and uncertainties (Costanza et al., 2017; Gómez-Baggethun et al., 2010; Schmeller, 2008; Scholes et al., 2008) and studies commonly undervalue ES. The undervaluation, for instance, presents forestry as less competitive and has enhanced the appetite for forestland conversion. Literature touts this as a primary cause of unsustainable exploitation, loss of biodiversity (Nahuelhual et al., 2007), and skewed budget allocations worldwide. This has ultimately led to a decrease in forest cover and a subsequent decrease in the supply and flow of ecosystem services.

This calls for further research aimed at unravelling the complex socio-ecological linkages (Howarth and Farber, 2002), and associated ES stock and flow (Fisher et al., 2008; Pascual & Muradian, 2010; Turner et al., 1994). The assessment of ES has commonly employed (stock) such as total amount of water in metric cubic per unit area (m^3/ha) and productivity such as the amount of water conserved in meters cubic annually (m^3/ha/year) (de Groot et al., 2010). While the research community has undertaken fewer studies on the ecosystem processes and functionality and the associated stock and flow of ES. In the policy context, the ES assessment provides an opportunity to show whether policies and policy interventions bring benefits or impose a cost on the ecosystem. Overall, ecosystem assessment and valuation will guide managers of ecosystems to identify high-value ecosystems (Alamgir et al., 2016; Baral et al., 2017). Besides provoking policy interventions to reverse degradation and support sustainable conservation. This will also broaden knowledge of ecosystem services (Mengist et al., 2020), raise the level of awareness, and demonstrate the importance of maintaining ES and how they contribute to the global agenda such as sustainable development goals (SDG) (Alamgir et al., 2014; Fagerholm et al., 2016). This will also form the basis for future policy development, especially the creation of market-based instruments, such as payment of ecosystem services (PES) (Christie et al., 2007; Engel et al., 2008). In addition, being a tool for environmental audit (DeFRA, 2007; Smith et al., 2013) and demonstrating the cost of environmental degradation (Pascual et al., 2010). It also provides evidence for improved resource allocation, presenting forestry as a competitive land use, and designing future conservation programs. This is besides awareness creation, and proposing policy interventions to provide alternative livelihoods, reduce pressure on state-owned forests, and improve conservation (Garekae et al., 2017; Gunatilake, 1998) of forested water catchments in the country.

2.2.2 ES Valuation techniques

Different valuation approaches and techniques exist, notwithstanding the shortcomings and uncertainties. Literature broadly groups the techniques as market-based and preference-based approaches. The market-based approach includes price-based (market pricing), cost-based (avoided, replacement, or mitigation/restoration cost), and production-based (productivity function approach and factor income). Preference-based techniques include the revealed and stated preference approach. Revealed preference includes hedonic pricing and travel cost, while the stated preference approach includes contingent valuation, choice model/ conjoint analysis, contingent ranking, and deliberate group valuation (Pascual et al., 2010). Each of the approaches records a wide range of pros and cons (de Groot, 2006) though the knowledge of assessment and approaches has grown over the years.

Despite the diversity of pricing techniques, price-based techniques adopt either direct or indirect market pricing. Direct market pricing includes the market price for goods and services bought and sold in commercial markets and their production function (Bertram & Rehdanz, 2013). However, literature commonly employs indirect market pricing for commodities not traded in the commercial market, such as the monetary value for wildlife refugia. The cost-based method equates the environmental value to the cost of replacing or restoring an ecosystem service (Balmford et al., 2002). While the preference-based method infers monetary units based on consumer preferences (revealed and stated) (Nijkamp et al., 2008; Remoundou et al., 2009). Revealed preferences, for instance, are the pricing of commodities based on observed consumer behaviour and commonly employ hedonic pricing and travel cost (Haab & McConnell, 2002; Pagiola et al., 2004). The hedonic pricing method serves to calculate the value of environmental goods such as landscape, air quality, and noise (Turner et al., 2010). This method evaluates the implicit price based on the individual willingness to pay for the relevant environmental characteristics. The environmental value is commonly based on the difference in commodity, such as house prices, time and site prices, and related expenses (Turner et al., 2010). The revealed preference techniques are useful only for use values, but stated preference techniques can assess both use and non-use values (Nijkamp et al., 2008) as well as the total economic value of an ecosystem. Stated preference techniques (Hajkowicz, 2007; Pagiola et al., 2004) derive ES values from individual responses and choices through hypothetical questions. For contingency valuation, individuals state the maximum willingness to pay to, for instance, maintain/improve the quality of the environment (Turner et al., 2010). While the choice experiment model presents options of ES with their relative value from which individuals choose (Hanley et al., 2001; Turner et al., 2010).

Studies employ a wide range of procedures to assess, quantify and assign monetary units for the respective ecosystem goods and services. For instance, direct market pricing would involve inquiring about the type of ecosystem services derived from the target ecosystem, the amount extracted, frequency harvest per unit, and relative unit price. While the indirect method includes observing the ecosystem status change and quantifying the impact on the stock and flow of ES and employing cost-based techniques to assign monetary units (Mäler et al., 2009). Studies commonly employ travel cost and contingent valuation techniques to determine direct non-consumable services, such as recreation and cultural activities. Direct use products with no defined market price such as fodder, game meat, and to some extent water use alternative product prices as surrogates. Net Primary Production (NPP) for instance, is among the various techniques used to estimate the value of ecosystem service as a proxy. The NPP is a product of many ecosystem functions and processes and, thus, will be a key attribute to measure nature's value. Though there is still a need for more data, the relationship between NPP and total economic value has been supported by literature, as it presents a simplified quantitative method (Costanza et al., 2014; Costanza et al., 1997).

However, the pluralist approach is more advocated for ES assessment though applied with precaution (Costanza et al., 2017). Most of these approaches are acceptable, as long as the research reports the uncertainties of each method used. An integrated and pluralist ecosystem assessment approach is critical to answering a wide range of conservation-related questions. This includes loss of biodiversity, global warming, and urbanisation, among others. This is contrary to the individualist assessment approach, which may not explicitly generate requisite information to guide future decision-making. The pluralist approach causes incorporation of both the cost and benefit of trade-offs, as well as establishing the relationship between them and how they influence societal welfare (Baciu et al., 2021). This approach will be critical for future ecosystem conservation policies and interventions.

Overall, data scarcity and gaps are one of the primary challenges in the quantification and adoption of some of these approaches. More need to be done to improve data collection, valuation approaches, and modelling in the estimation of ecosystem services either as a unit or aggregate value (de Groot et al., 2010).

2.2.3 Typology and TEV Framework

Ecosystem services classification provides a mechanism for the assessment and valuation of biodiversity and ecosystem resources. Despite the ambiguity in the terminologies, it is important to classify ecosystem services to allow stakeholders to make comparisons and evaluate the consequences of different management actions, policies, and policy interventions (Wallace, 2007). Based on product utility, this will also guide the assessment, valuation, comparative analysis, and modelling of ES (Costanza et al., 2017). The classification of ES is critical in guaranteeing simple assessment and assignment of monetary units to ES (Englund et al., 2017; Fisher et al., 2009a; Fu et al., 2010; Nemec & Raudsepp-Hearne, 2013). Notably, different techniques and approaches exist in the literature for ecological services classification (Burkhard et al., 2010; Häyhä et al., 2015; Paudyal et al., 2015). The selection of the typology is principally based on the type of ecosystem services, study scope and scale, data requirement and availability, cost time, and expertise (Baral et al., 2017). However, the inconsistencies and pluralism in the classification of ecosystem services make it difficult to integrate such diverse findings in a meta-analysis (Haines-Young & Potschin, 2011).

Despite the different ES typologies, researchers, however, have commonly adopted the total economic valuation (TEV) framework. This is primarily because of its alignment of ecosystem services based on utility, regardless of the applied ES typology. The total economic valuation (TEV) is the summation of appropriately discounted values across ES typologies. TEV groups ES as use and non-use (de Groot et al., 2010; Krutilla, 1967; Pearce & Turner, 1990) and also disaggregated it into different components of values (Balmford et al., 2008; de Groot, 2006; de Groot et al., 2002) as shown (Figure 2.3). The framework integrates the ecological, economic, and socio-cultural dimensions of such assets (Farber et al., 2002; Howarth & Farber, 2002; Limburg et al., 2002; Wilson & Howarth, 2002).

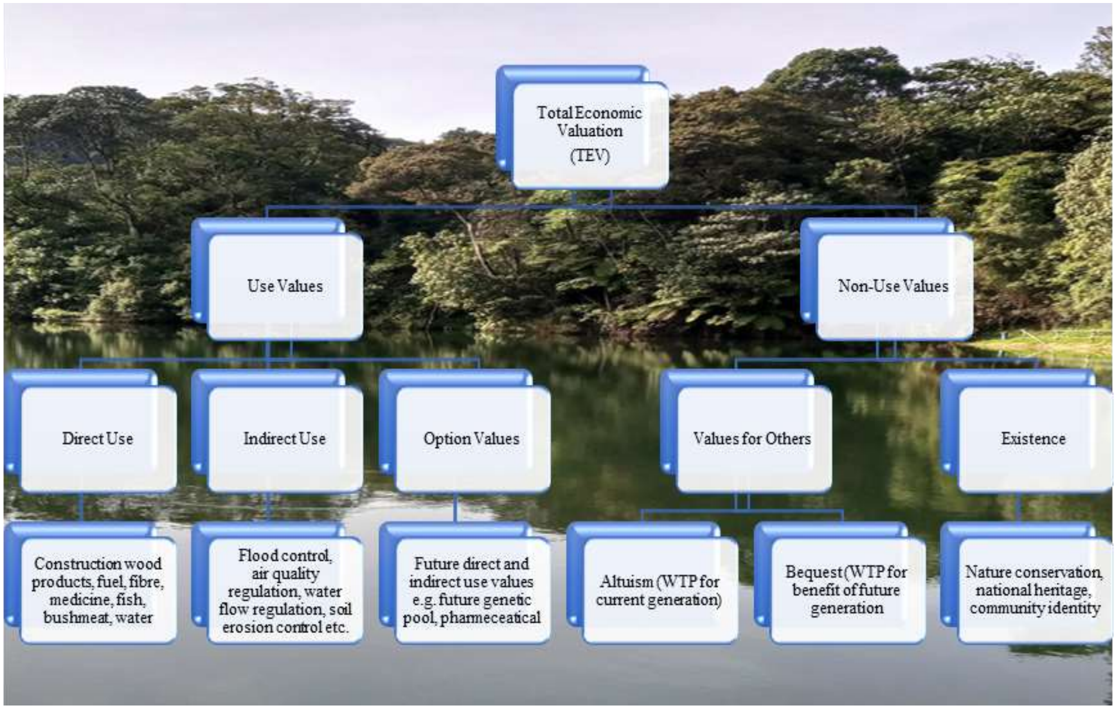

Figure 2.3:Total Economic Valuation (TEV) Framework

source; (Martín-López et al., 2009)

Studies commonly report ES assessment estimates either as a unit or aggregate value. The total, in this case, does not imply an absolute figure but captures part of the aggregate or 'true' value of the ecosystem. Though the research could not meet the theoretical 'whole' ecosystem value, the aggregated estimates of the subset ES assessed are what studies would report as total economic value (Aboud et al., 2012; East Africa Commission (EAC), 2014; Nahuelhual et al., 2007). This section below explains and reports the studies on ecosystem services based on the TEV framework (Figure 2.3).

2.2.4 Direct use values

Direct use values are categorised as consumptive products, for example fuelwood, water, and non-consumptive/non-extractive use, such as appreciation of landscape, aesthetic, and spiritual values. Consumptive values have a market price since traded in the conventional market, while non-consumptive has an indirect market value (DeFRA, 2007; Pascual et al., 2010).

2.2.4.1 Direct Consumptive use values

Consumable direct-use products include timber and non-timber forest products (NTFP) though more commonly reported in national economic discourse than the latter. The non-timber forest products are equally important to societal welfare. The NTFP are all tangible forest products other than timber, collected for subsistence and as a source of income (Ros-Tonen et al., 1995; Ros-Tonen & Wiersum, 2005). The non-timber forest products include plants and plant materials used as food, fuelwood, livestock fodder, fibre, and biochemicals, among others (Mainga, 2016). The respective economic and consumptive values attached to them (Duong, 2008) motivate their extraction. Kumari (1996) estimated the value of ecosystem protection and providing water for irrigation at USD 15 ha^{-1} yr^{-1} (Nahuelhual et al., 2007). While a study in Indonesia estimates the value of provisioning services in a community-owned oil plantation at USD 4,331 $ha^{-1}yr^{-1}$ (Aulia et al., 2020).

2.2.4.2 Direct Non-consumptive Use

Non-consumptive use values are ecosystem benefits drawn based on the societal norms and beliefs that bear identity and diversity (Bingham et al., 1995; Farber et al., 2002; Fisher et al., 2009b). Studies commonly express the non-consumptive use values in terms of activities, practices, and happenings that occur in the forest (Tabbush, 2010) such as nature walks (Kreye et al., 2017). They include services such as recreation, aesthetics, education, justice, freedom, and spirituality, among others. Despite the appreciation of non-consumptive use values, society has not sufficiently defined and assessed non-use values in the ES framework (Daniel et al., 2012). Literature attributes this to uncertainty on their association with, for instance, artifacts and monuments and the suitability of the valuation methods (de Groot et al., 2005). Forested ecosystems support nature-based recreational activities through the provision of beautiful sceneries, topography, water flows, flora, and fauna (Ceballos-Lascura, 1996). Valuation of non-use values commonly uses revealed, for example travel cost and stated preference, such as contingency valuation. Some countries used the results of such studies in determining and revising conservation gate fees, concessions, and appropriation of the national budget for protected areas (Bishop, 1999).

In Kenya, records of game-driven nature-based tourism are in National Parks and Reserves which host diverse fauna targeted by game-driven foreign tourism. The local tourist, however, would also visit forest ecosystems to appreciate the beautiful landscape, social gathering, and spiritual connections without game animals similar to the case of the Elgeyo and Nyambene. In Chile, estimated nature-based recreational value using travel costs at USD 6.3 and 1.6 per hectare annually for Puyehue and Perez Rosales national parks, respectively (Nahuelhual et al., 2007). Though the Puyehue and Perez recreational value only considered national tourism in public areas, the estimates were lower than the tropical forest recreational value, reported at USD 2000 per hectare annually. However, the values were higher than the European temperate and boreal forests, estimated between USD 0.5 and 3 ha^{-1} $year^{-1}$ (Pearce, 2001). The valuation of recreational values should be sensitive to the disaggregate values, the ecosystem, season, and the number of visitors incorporated into the survey since is likely to underestimate the economic value. If sensitivity analysis would consider some of these factors, the recreational values are higher than reported (Nahuelhual et al., 2007). The estimate of the cultural services (sacred sites) in a community-owned oil plantation in Indonesia was about USD309 $ha^{-1}yr^{-1}$ (Aulia et al., 2020).

2.2.4.3 Indirect Use Values (IUV)

Indirect use values make up regulatory and support services based on the Millennium ecosystem assessment typology (Millennium Ecosystem Assessment (MA), 2005). The literature described them as products of ecosystem functions and processes that benefit humanity indirectly. These services include climate regulation, regulation of air quality, soil erosion control, soil nutrient conservation, and cycling and pest and disease control, among others. Over the years, society has taken for granted these regulatory ecosystem services. This continues until when the level of degradation and impact now becomes apparent and, in most cases, it becomes difficult to restore them (TEEB, 2012). The literature classifies regulatory ecosystem services (RES) based on their interaction and their effect on humanity, either as final or intermediary RES (Watson et al., 2011). For instance, it classifies pest and disease control, nutrient cycling, and seed dispersal, among others, as intermediary RES, since they affect humanity indirectly (Mengist et al., 2020). While soil conservation, water purification, climate regulation (Smith et al., 2013), and flood control (Stürck et al., 2014) are the last category of services because they affect humanity directly (Mengist et al., 2020).

Equally, state and non-state actors have ignored indirect use values in national accounting and development despite their immense contribution (Kandziora et al., 2013; Sutherland et al., 2018). Literature attributes this to complexity in linking human benefit with the indirect RES (Mengist et al., 2020), low understanding of the link to human well-being, and a lack of market price for most of them. This is in addition, the complexity in the detection of marginal change may go unnoticed until it surpassed the ecosystem thresholds (DeFRA, 2007). Globally, a significant focus is on direct use values (Sutherland et al., 2018; Villamagna et al., 2013). This poses a challenge in generating adequate data that explicitly accounts for total monetary values (Villamagna et al., 2013). In that regard, research should endeavour to account for both direct and indirect use values to inform the policy and decision-making processes adequately. Valuation excluding subsets of ES would undervalue and cannot show explicitly the ecosystem's worth and the relative trade-offs (Keeler et al., 2012). Studies commonly employ the 'shadow pricing' approach for the valuation of RES, since conventional markets do not trade them (Barbier, 2007; Morse-Jones et al., 2009; Polasky & Segerson, 2009). This is besides the lack of clear market prices (Scheierling et al., 2006; Young, 2005) as explained here below.

2.2.4.3.1 Watershed Regulation function

The water catchment ecosystems act as "a green" reservoir because of its immense 'osmosis' effect with the ability to capture and store precipitation and discharge it through various water sources such as springs, and rivers, among others (Guo et al., 2000; Wu et al., 2010). These functions manifest through the ability of an ecosystem to intercept, trap, and keep precipitation through its crown, trunk, undergrowth vegetation, forest litter, and soil. This function supports filtration, retention, and the continued flow of water in reservoirs (Nahuelhual et al., 2007) based on the structure and characteristics of the ecosystem (de Groot et al., 2002). Rainfall interception by the vegetation canopy is directly proportional to both the leaf area index (LAI) and rainfall intensity discharge. The leaf interception ranges from 11% to 22.8% of total precipitation in tropical and temperate forests and 8% to 18% in tropical agricultural land (Cuartas et al., 2007). The precipitation interception by litter is less than 20% of the total precipitation (Acharya et al., 2016; Dunkerley, 2015) though in isolated cases, that can record as high as 70% of the total precipitation (Dunkerley, 2015). Through these processes, forested ecosystems recharge groundwater supplies, maintain base-flow stream levels, and lower peak flows during heavy rainfall or flood events (McGuire, 2009; WRI, 2011). Enhancing springs and river flow and continued water flows over time, in the dry season (Guo et al., 2000; Ninan, 2011; Pike et al., 2010).

The structure of an ecosystem influences the water flow regime, characterised by base flow, seasoning timing, variation, and frequency (Binder et al., 2017). In that regard, degradation and conversion of forested ecosystems to other land uses interferes with the hydrological system (Ataroff & Rada, 2000; Calder et al., 1997;

Putuhena & Cordery, 2000). This includes interference of flow regime and water supply, which leads to extreme base and peak flow, more pronounced in dry and rainy seasons (Guzha et al., 2018; Qazi et al., 2017). A functional ecosystem indirectly enhances the availability of fresh water, preservation, and maintenance of aquatic life, riparian ecosystem, and wildlife habitat among others.

Similarly, the forest ecosystem provides clean water from non-point sources through the process of natural filtration and improves water quality (Pike et al., 2010). It manifested this through the interaction of precipitation water with vegetation and the stabilisation of soils through root systems. Many global forests play a key role in trapping, filtering, stream flow generation, routing, and absorbing contaminants. This ecosystem function regulates water chemical levels besides modifying water chemistry (Bamlaku & Tesfay, 2015; Jahanifar et al., 2017; Krieger, 2001; Stednick, 2010). Equally, it improves water quality through the additional valuable forest minerals and nutrients that are good for human health (Ninan, 2011). In that regard, any form of disturbance to the ecosystem would alter this function.

Valuation of regulatory ecosystem services can be based on the ultimate benefit to humans. The production function method can estimate the indirect use values, such as river discharge, primarily as a proxy for ecosystem structure and functionality (Maller 1992). Studies commonly employ shadow prices to assign a monetary value to this function. For instance, when using the water regulation function on flood risks often apply the cost of damage avoided, with the assumption that those properties downstream are likely to be affected in an event of flooding. In that regard, studies commonly use the value of properties downstream as a shadow price for the ecosystem flood control function. Though the concept seems straightforward, the actual assessment and implementation are complex. However, studies commonly use mathematical models to predict the extent and magnitude of flood risk in a water flow based on land use change (Watson et al., 2016). In addition, using historical experience, the model can estimate the extent and intensity of the damage to properties and the associated cost (Binder et al., 2017).

According to a study in Chile, the economic value for water regulation function for native Chilean forest range between USD 61.2 and 162.4 ha^{-1} yr^{-1} based on production function (Nunez et al., 2006). Equally, a study in Brazil by Torras (2000) estimated water regulatory services contributed by the Amazon forest at USD19 ha^{-1} yr^{-1}. While a study in Zagros Forests, Iran estimated the water storage function forest ecosystem at USD 43 ha^{-1} yr^{-1} (Mashayekhi et al., 2010). Whereas the unit value for aggregated regulatory services (groundwater recharge, carbon sequestration) in a community-owned oil plantation in Indonesia was USD 1,880 $ha^{-1}yr^{-1}$. The study equally valued groundwater recharge, at USD 1806 $ha^{-1}yr^{-1}$, accounts for 96% of the aggregated regulatory services value (Aulia et al., 2020).

2.2.4.3.2 Soil and nutrient conservation

Soil anchors most of the life forms on earth, and its critical in global growth through the support of critical economic sectors such as agriculture energy, transport, and forestry, among others. However, deforestation and degradation expose forest soils and consequently hasten precipitation runoff, resulting *in situ* soil erosion (Nahuelhual et al., 2007; B. Zhang et al., 2015) and *ex-situ* sedimentation. This is besides loss of soil nutrients, deterioration of water quality, eutrophication (He et al., 2003), and interference with the hydrological cycle (Adger et al., 1995). The off-site sedimentation leads to reduced capacity and lifespan of the water reservoirs (Mahmood, 1987; Schleiss et al., 2016) and eventual abandonment (Kawashima, 2007). However, soil degradation associated with erosion threatens the very life and developments that depend on it (Dotterweich et al., 2013; Gebrehiwot et al., 2014).

Research has recorded huge soil loss within arable areas in the last four decades, estimated at over ten million hectares annually, equivalent to a third of the productive area globally (Pimentel et al., 1995). This has affected land productivity negatively and food security more so in developing countries (Tully et al., 2015; Wynants et al., 2019). The major agent of soil erosion is water and wind, with the respective effect being more pronounced in bare and degraded ecosystems. The anthropogenic actions and practices (B. Zhang et al., 2015) have exacerbated this natural phenomenon. Soil conservation strategies include forest conservation, artificial soil conservation approaches and techniques, and soil erosion-proof agricultural practices, among others (Kuhlman et al., 2010). Soil nutrient level depends on soil structure that is relatively influenced by landscape climate, topography, vegetation cover, and root structure (de Groot et al., 2002). Water catchment ecosystems, such as the Elgeyo and Nyambene, play a critical role in soil stabilisation by reducing the direct impact of precipitation. This is primarily associated with forested ecosystems' leaf area interception and root structure holding ability (de Groot et al., 2002).

Studies have estimated the amount of soil lost through erosion to range from 1 to over 500 tons ha^{-1} $year^{-1}$, globally (Endalamaw et al., 2021; Eshghizadeh et al., 2018; Kateb et al., 2013; Shrestha, 2016; Tessema et al., 2020). Principally, this depends on the land cover type, vegetation type, slope, soil type, and precipitation intensity. Undisturbed native forest loses the least amount of soil annually, less than 0.1ton ha^{-1}. However, when disturbed through clearing and conversion to other land uses, it can lose up to about 300 tons of soil per hectare annually. Nutrient loss is relative to the amount of topsoil lost through erosion. A study by Mancilla (1995) recorded soil mineral losses at 1.26kg ha^{-1} $year^{-1}$, 0.82 kg ha^{-1} $year^{-1}$, 0.14 kg ha^{-1} $year^{-1}$, and 15.13 kg ha^{-1} $year^{-1}$ for potassium (K), nitrogen (N), phosphorus (P) and calcium (Ca) respectively (Mancilla, 1995).

The valuation of ecosystem soil nutrient conservation is determined using the replacement cost approach. This assumes the loss of soil fertility would require the application of an alternative (Bishop, 1999) in this case commercial fertiliser to replace soil nutrient loss (Nahuelhual et al., 2007; Riera & Signorello, 2013; Turner et al., 1994). In that regard, studies commonly use the cost of commercial fertilisers as a surrogate for soil nutrient conservation function. This is primarily determined based on relative nutrient-fertiliser ratios reported by the respective commercial fertiliser producing companies.

A study conducted on Chilean temperate forests estimated the economic value for ecosystem soil fertility protection at USD 26.3 ha^{-1} $year^{-1}$ (Nahuelhual et al., 2007). A study by Bann estimated soil fertility at USD46 ha^{-1} $year^{-1}$ (Bann, 1998), while a study in tropical Amazon forest estimated soil fertility value at USD130 ha^{-1} $year^{-1}$ (Torras, 2000). The value of soil fertility varies across ecosystems largely explained by the susceptibility to soil loss influenced by the type of vegetation, soil structure, and the annual amount of precipitation (Nahuelhual et al., 2007).

2.2.4.3.3 Climate Regulation

Terrestrial ecosystems such as forests play a critical role in global climate regulation as they sequester a significant amount of CO_2 that has high warming potential (Pan et al., 2011; Scharlemann et al., 2014; Zheng et al., 2008). Equally, forest soil is a major carbon sink and contributes significantly to climate change mitigation if sustainably conserved and managed (Lal, 2004, 2005; Maher et al., 2010; Mishra et al., 2019). Forest also plays a critical balancing role of oxygen and carbon dioxide gases in the atmosphere through the capture and storage of CO_2 and release of O_2 as a bio-product. Assessment of forest plant and soil carbon is important to monitor fluxes in atmospheric CO_2 as well as provide requisite data for the growing interest in reducing carbon emissions (Campbell et al., 2009; Sayer et al., 2019).

Atmospheric CO_2 concentration has almost doubled from the industrial period level of 280ppm to over 402ppm as of 2016 and increasing by 1.5ppm annually (Kencky & Pieter, 2016). The drastic increase in atmospheric CO_2 has been linked to anthropogenic drivers, including the combustion of fossil fuels and the conversion of carbon sinks and reservoirs such as forests ecosystem to other land uses (IPCC, 2007, 2014; Shitanda et al., 2013). The land-use change, for instance, accounts for approximately 20% of the increase in CO_2 and consequent soil erosion contributing between 50–70% loss of organic carbon (Gibbs et al., 2007; Houghton & Nassikas, 2018; IPCC, 2018; Lal, 2003). Carbon lost through the conversion of forests to other land use change in Brazil, Indonesia, and African equatorial forests reporting between 100 and 150 tonnes per hectare annually (Guo, 2002; Hairiah et al., 2001; Palm et al., 2001).

Terrestrial ecosystems, particularly tropical forests, store more than 400Pg of carbon with more than half as soil organic carbon (Pan et al., 2011; Scharlemann et al., 2014; Zheng et al., 2008). Temperate forest soil carbon accounts for almost double vegetation carbon, while some forest ecosystems SOC make up more than half of the total terrestrial forest carbon. This is a sign that forest soil is the most important and active carbon sink in the global carbon cycle (Lal, 2004, 2005; Maher et al., 2010; Mishra et al., 2019). Similarly, forest soil carbon is critical in enhancing forest productivity, including above-ground biomass, that is higher SOC correlates with tree biomass and vice versa. Forest degradation, deforestation, and unsustainable agricultural practices expose carbon sinks, which then lead to the emission of GHGs such as CO_2, CH_4, N_2O, H_2O vapour, and industrial gases into the atmosphere, thus contributing rise in global temperature. Globally, forest ecosystems lost over six million hectares to deforestation and degradation, which translates to tonnes of CO_2 released into the atmosphere (IPCC, 2014). Assessment of above-ground and below-ground carbon is important as society endeavours to monitor fluxes in atmospheric CO_2 as well as provide requisite data for the growing interest in reducing carbon emissions (Campbell et al., 2009; Sayer et al., 2019).

Primarily, In Kenya, more focus of the forest industry has been on enhancing standing stock carbon biomass and not soil carbon conservation *per se.* Stakeholders in the industry are in consensus that how we manage our forest ecosystems has a direct impact on soil carbon and global carbon balance (Bradley & Scott, 2011; Vesterdal & Leifeld, 2007). Carbon stock assessment in Kenya would be important for the establishment of the country's carbon emission and reduction baseline. This will provide a platform for exploring REDD+ initiatives and carbon trade markets (Eregae et al., 2016). Carbon sequestration is of interest not only nationally but to the global commitment to the reduction of CO_2 emission (Rawlins et al., 2008). Article 4 of the UNFCCC, calls for enhanced efforts in eliminating anthropogenic-led greenhouse emissions through the protection and enhancement of sinks and reservoirs worldwide (UNFCC, 2006).

The amount of forest CO_2 captured and O_2 generated is based on the amount of biomass, in this case, using the improved pan allometric algorism (Chave et al., 2014) and photosynthesis process formula (2.1).

$$6CO_2(264g) + 12H_2O\ (180g) \rightarrow C_6H_{12}O_6(180g) + 6O_2(192) \qquad 2.1$$

From the equation above, using the atomic mass of the respective elements, the ratio of one-ton carbon dioxide sequestered to the oxygen released at a ratio of 1:0.73.

Valuation of carbon commonly employs market pricing, though some literature argues that the technique doesn't reflect the true value and recommend a climate change damage function (Ferarro et al., 2011). Currently, carbon credit prices vary from a few cents to over $30/MgCO2e (Vanlandingham, 2021). However, for the case of the Kenya REDD+ project carbon pricing, currently retailing at USD 5, though they are negotiating for higher prices of about USD 15. In that aspect, and based on the market pricing technique, the economic value for carbon sequestration in a community-owned oil plantation in Indonesia was USD147 $ha^{-1}yr^{-1}$ (Aulia et al., 2020).

Notably, two carbon market mechanisms exist globally, namely regulatory compliance and voluntary market mechanism (Hamilton et al., 2010). The two market mechanisms differ, though they both contribute to the global market. By 2021, the global carbon market made up about $280 billion, with the European Union's Emissions Trading System (EU ETS) accounting for approximately 90% of the global market value (Reuters, 2021). This has since grown with the involvement of countries such as China that committed to enhancing its budget in the carbon market. Equally, worth reading is the Paris proposed models, that is the "contribution claim", "Nationally Determined Contribution (NDC) crediting" and "non-NDC crediting". The NDC crediting, for instance, requires the respective countries to set mitigation targets, authorisation, corresponding change, and neutrality claims. This then calls for the redesigning of voluntary carbon market models to facilitate and support governments' NDC climate mitigation efforts (Fearnehough et al., 2020). Likewise, the Carbon Offsetting and Reduction Scheme for International Aviation (CORSIA). This is besides other emission reduction options, such as the purchase of voluntary carbon credits and liquified natural gas "carbon neutral" energy initiatives.

The voluntary carbon market, for instance, has a huge potential to support and facilitate sustainable conservation (Vanlandingham, 2021).

2.2.4.3.4 Microclimate influence

Categories of climate include micro, macro, synoptic, and mesoscale based on their spatial and temporal attributes (Hauk et al., 2012). Forest ecosystems enhance local moisture, temperature, soil structure, and nutrient, among others, thus creating a forest-related microclimate (Hauk et al., 2012; Kurtural et al., 2007). Forest microclimate includes understory and forest edge effects (Chen et al., 1999). The understory effect, for instance, develops from the forest shadowing effect influenced by radiation, while the forest edge effect influences the forest itself and bordering environs (Kurtural et al., 2007; Norris, 2012; Wright et al., 2010). Changes in air temperature chiefly influence this variation in the ecosystem (Medellu et al., 2012). Literature attribute this kind of climate to the ecosystem floral structure (Yu & Hien, 2006) and microscale atmospheric circulation (Hauk et al., 2012). This is besides ecological and physiological processes and functionality fundamental for tropics classification (Brovkin, 2002). According to Spangenberg et al. (2008), trees have a higher influence on microclimate than the lower level vegetation, such as shrubs and grass. Though different vegetation records distinct microclimates (Smith & Johnson, 2004), forest ecosystems record enhanced microclimate influence over other classes and structures of vegetation (Muhlenberg et al., 2012; Von Arx et al., 2012). According to Manyanya and Kori (2014), forest microclimate influences agriculture production, where the study recorded a significant decrease in crop yields with an increase in the distance from the forest edge. Therefore, agriculture production changes are a response to variations in microclimate intensity. According to Kipkoech et al. (2011), forest edge microclimate contributes between eight and twenty percent of agricultural production.

2.2.4.3.5 Insect Pollination

Globally, approximately 35% of agricultural production is reliant on animal pollinators (Klein et al., 2007) and critical for food security and socio-economic development, particularly for small-scale farmers (IPBES, 2016; Picanço et al., 2017). Wild pollinators are important for the natural regeneration and resilience of the host forests and enhanced crop yields and quality for farmland bordering forested ecosystems (Nitharwal et al., 2021; Sawe et al., 2020). They are also critical in providing necessary calories and micronutrients to humanity (Sundriyal & Sundriyal, 2004). Forested ecosystems play a critical in hosting wild pollinators as they act as nesting and foraging areas which vary across diverse ecosystems (Krishnan et al., 2020). This is the case since a wide range of crops and wild plants are reliant on animal pollinators and their value varies significantly across landscapes and regions (IPBES, 2016). It also linked the decline in plant diversity to a decline in natural pollinators (Biesmeijer et al., 2006).

Overall, animal pollinators play a pivotal role in plant development and crop productivity. An abundance of pollinators would primarily enhance both the quantity and quality of production *in situ* and *ex-situ*, more so on perennial crops (Gallai et al., 2009; Klein et al., 2007). According to Kasina & Kitui (2007), global food production is reliant on pollination functions either by natural agents or artificial agents. However, land use change, unsustainable agricultural practices, human population pressure (Picanço et al., 2017), and high demand for land food production have impacted negatively natural pollinators (Marshman et al., 2019). The extensive application of conventional pesticides exacerbated the situation (Picanço et al., 2017). Literature touted the loss of natural pollinators as the major contributor to the decline in on-farm productivity and global food security (Marcelo et al., 2009). That notwithstanding, farmers next to a biologically diverse ecosystem reap the benefit of natural pollinators and reduced crop pests infestation (Bianchi et al., 2006) through the natural pest control mechanism.

Pollination assessment is primarily used to estimate the indirect economic contribution to agricultural production, livelihood, and economy (Sawe et al., 2020). This will also show the cost of losing pollinators in nutritional security, livelihood, and food security (Hwalla et al., 2016; Smith et al., 2015). Monetary valuation of animal pollination would require a holistic approach to incorporate its value into the different aspects of livelihoods. Otherwise, valuation based on marketed products alone would undervalue the societal benefit of animal pollination (Klein et al., 2007).

In that regard, the contribution of pollination, for instance, to crop production should consider its value at a local or regional scale to illustrate value at a small production unit, that is household farm. Further analysis should also assess the contribution of animal pollination to nutrition and food security (Hwalla et al., 2016; Smith et al., 2015).

2.2.5 Option value

Despite the critique of their incorporation of option values into total economic valuation (Freeman, 1993), appreciation of options use values is critical for maintaining and preserving nature for future use (Krutilla & Fisher, 1975). Literature also referred the latter to as quasi-option values (Pascual et al., 2010). An example of option value includes when the stakeholders invest to preserve and maintain an ecosystem even though they are presently not benefiting but recognise its future benefit. Equally can appreciate option use values based on ecosystem insurance value (Balmford et al., 2002; Turner et al., 2003) for example preservation of biodiversity would ensure ecosystem resilience in case of perturbation in the future (DeFRA, 2007).

2.2.6 Non-use value

Literature connects passive use values to intrinsic environmental values based on the knowledge of the existence of an ecosystem (Pascual et al., 2010). They comprise bequest (preservation and maintenance of nature for future generations), and altruism (preservation of the ecosystem for other people in the current generation). This is besides existence values (preservation without benefiting from it, such as the Queens project willing to invest in an African forested water catchment without evening knowing the actual location of the forest). Valuation of non-use services is even more difficult because these ES are not traded in conventional markets. Because of the unavailability of market price, stated preference has commonly been used to assign a monetary unit to this category of services. Studies attribute these values to the societal willingness to pay to preserve the ecosystem and related ES. Ecological, social-cultural, and economic attributes influence the monetary value assigned and the maximum willingness to pay (Barbier et al., 2009; Pearce, 1993).

2.2.7 Total Economic Valuation (TEV)

ES assessments commonly report monetary estimates as a unit, aggregate value, or both. The aggregated values encompass all aspects of utility and preferences drawn from the subject ecosystem and translated into monetary terms (TEEB, 2010a). while the unit value is based on the average aggregate value per unit area. According to Costanza et al. (2014), the aggregate global nature value ranges between USD 125 to 145 trillion. While a locally based study in the Mazandaran forest reserve estimated the aggregate ES value at about USD 14.5 million, an equivalent of USD 6,817.97 ha^{-1} yr^{-1} (Jahanifar et al., 2017). The assessment of community-owned oil plantations in Indonesia valued ES at USD 6,520 $ha^{-1}yr^{-1}$ (Aulia et al., 2020). While Nepal community forest reported a unit area value of USD 2265 ha^{-1} yr^{-1} (Dhungana & Deshar, 2019). Virginia state forest reported a total economic value of USD 15.3 billion equivalent to USD 2,175.36 $ha^{-1}yr^{-1}$ (Paul, 2011). While Florida State Forest reported a discounted aggregate value of 2.07 billion equivalent to USD 372.97 $ha^{-1}yr^{-1}$ (Escobedo & Timilsina, 2012). Likewise, a study on private forest land in Georgia estimated the aggregate ES value at USD 37 billion, an equivalent of USD 4,221.23 $ha^{-1}yr^{-1}$ (Moore et al., 2011). Equally, a study in the Texas state forest reported a value of USD 92.9 billion, an equivalent of USD 3,677.83 $ha^{-1}yr^{-1}$ (Simpson et al., 2013). These studies exhibit a wide range of monetary values, from about $372 in Florida to $4,221.23 $ha^{-1}yr^{-1}$ in Georgia is a demonstration that the ES economic values vary across landscapes and regions.

Literature principally attributes the variance in both aggregate and unit values to valuation methods, approaches, techniques, and the study scope. The study in Florida, for instance, only incorporated use values, while Texas and Georgia also included non-use values besides use values. Here, all four studies used benefit transfer but still recorded different values (Richardson et al., 2015). Some studies would choose a single study value, while others would consult a wide range of studies and compute a mean as a unit value and extrapolate to the entire ecosystem. Equally, the supply, demand, and purchase power vary across regions and landscapes, and even when using a similar technique and approach, it may still record variance (Sills et al., 2017). The approach can also lead to variation in economic value, where some researchers would treat an ecosystem as homogenous while others assume a heterogeneous landscape, thus variance in values becomes inevitable.

2.2.8 Forest Dependency

Globally, forested ecosystems support over 1.6 billion people (World Bank Group, 2002) through the provision of both wood and non-wood forest products (Angelsen et al., 2014; Babulo et al., 2009; Balmford et al., 2002; FAO, 2010; Fikir et al., 2016; Shackleton et al., 2007). They are critical, particularly in low-income forest communities (Bwalya, 2013; IUCN, 2001; Mukul et al., 2016; Shackleton & Shackleton, 2004; Uberhuaga et al., 2012). These ecosystems not only act as a source of household livelihood (Najabat et al., 2020) but also provide opportunities for employment, income, and economic growth (Babulo et al., 2009; Bahuguna, 2000; Kabubo-Mariara, 2013; Mamo et al., 2007; Shackleton et al., 2007). They are equally a source of nutrition and natural medicine (Murkherjee & Chaturvedi, 2017) critical for human well-being. In Sub-Saharan countries, for instance, forests make up approximately 40% of the total household income (Appiah et al., 2009; Kalaba et al., 2013; Mamo et al., 2007) and about 10% to 20% for Asia and America, respectively (Córdova et al., 2013; Mukul et al., 2016; Uberhuaga et al., 2012).

Many of the forest-adjacent communities in developing countries such as Kenya depend on both timber and non-timber forest products (NTFP) for livelihoods (Heubach et al., 2011; Jana et al., 2017; Jarernsuk et al., 2015; Ndoye et al., 1998; Sumukwo et al., 2013). According to a study in the South Nandi forest, approximately 55% of the rural population spent most of their time collecting these products (Maua et al., 2019). It reported the harvest and sales of wood products to have doubled in the last decade, from about 400 thousand to a million metric cubic (Gichora et al., 2010). Literature value the NTFP harvest at about USD 579.51 per household annually. While livestock grazing value ranged from USD 12 to 205 per household annually (Maua et al., 2019). Overall, NTFP contributes approximately 40% of the household income (Heubach et al., 2011; Maua et al., 2019). Equally, according to KFS (2014), timber harvest alone contributed about 3.6% of the country" gross domestic product (GDP) in 2012. Besides these ecosystems contribute approximately 80% of Kenya's energy hydroelectric power (HEP) (MoE & F, 2004). Overall, these ecosystems support critical economic sectors such as agriculture, manufacturing, trade, and tourism (MEWNR, 2013; MoE & F, 2004; Shackleton et al., 2002). This depicts their importance to the forest community's livelihood and national economy (Dovie & Witkowski, 2002; Godoy & Bawa, 1993; Nahayo et al., 2013; Ticktin, 2004).

The forest-adjacent communities are heavily reliant on forest resources as a source of income and employment (Muthike, 2016; Nahayo et al., 2013; The Republic of Kenya, 2018; Ticktin, 2004). However, community dependency on forest resources varies across geographical location, socioeconomic status, time, and culture, among others (Babulo et al., 2008; Bhavannarayana et al., 2012a; Bwalya, 2013; Panta et al., 2009a). The study attributes this to the heterogeneity of the forest-adjacent communities in terms of socioeconomic and cultural traits (Coomes et al., 2004; Córdova et al., 2013; Langat, 2016; Sapkota & Oden, 2008). The forest community's diverse socio-economic traits influence the level of extraction and utilisation (Garekae et al., 2017; Kalaba et al., 2013; Langat et al., 2016; Najabat et al., 2020). Thus, making any inference and prognosis on forest dependency is a function of socio-economic traits (Bhavannarayana et al., 2012b; Panta et al., 2009b). This includes the length of residency, household size, sex, education, income, and livestock size, among others (Garekae et al., 2017; Kalaba et al., 2013).

According to Garekae et al. (2017), the age of forest users can inversely influence the score of forest dependency, contrary to the norm where forest harvest is associated the older folks (Adam & El Tayeb, 2014; Htun et al., 2017; Mujawamariya & Karimov, 2014; Ofoegbu et al., 2017; Suleiman et al., 2017). This is a significant shift where young people exhibit high forest dependency than older folks. Equally, education inversely influences forest dependency negatively, where the lower level of education records a high score on forest dependency and vice versa. Other critical parameters that are likely to influence forest dependency include the length of residency, household size, sex, distance from the forest, land size, livestock, land use, and employment (Garekae et al., 2017; Gatiso, 2017; Najabat et al., 2020).

2.2.9 River Flow Dynamics and Effects of Land Use Changes

Globally, metrological and anthropogenic factors influence river hydrological dynamics (Langat et al., 2019). The changes in river geomorphological processes, influences the availability of water, flood risks, vegetation structure, and river water chemistry (Milly et al., 2002; Pumo et al., 2013). Metrological dynamics, ecosystem structure, and ecological functionality keep on changing based on natural and anthropogenic forces (Lane et al., 2010; Piégay et al., 2015). Though society has little to do with metrological regime change, there is more that needs to be done on the anthropogenic front. Studies link the decline of ecosystem functionality to anthropogenic pressure, coupled with climate change. Conversion of forest land, for instance, has contributed to river flow and hydrological system change (Kigira et al., 2008; Mutie et al., 2006). Subsequently, the river basins have experienced extreme hydrological regimes (Fuller et al., 2003; Yang et al., 2002) though at varying scales across river basins (Petit et al., 1996; Surian, 1999). This has resulted in faster flow, decreased lag period, and enhanced peak flow in rainy seasons. Likewise, reduced base flow and deterioration of water quality which is associated with enhanced surface run-off and sedimentation (Chang, 2007; Dougherty et al., 2007). This has exacerbated the already scarce fresh water globally and, as a result, increased water demand and the subsequent water-related conflicts (Cosgrove & Rijsberman, 2000; Taha, 2007). The declining ecosystem has led to an inability of the ecosystems, for instance, to regulate water flow, soil conservation, and climate regulation among other services (Saleh et al., 2013). The last two decades have recorded immense water demands and the subsequent development of mega water reservoirs. This has also resulted in the illegal diversion of river water upstream, which has affected aquatic life and downstream human population (Kibet et al., 2019; Kohler et al., 2015).

The major river basins in Kenya have recorded water conflicts and a decline in riverine river flows (Hamerlynck et al., 2010; Saenyi & Chemelil, 2003) and subsequent alteration of the river flow system (Leauthaud et al., 2013; Maingi & Marsh, 2002; Mutisya & Mutiso, 1998). Primary assessment attributes this establishment of mega dams to the river channel, such as the Ura River in Nyambene with about four mega dams established along its channel. This, for instance, has largely contributed to the alteration of tropical rivers' hydrological regimes in major river basins in Kenya. Equally, the literature associates land cover and use changes with hydro-morphological river regime change (Kibet et al., 2019; Kitheka et al., 2019; A. N. Mwangi et al., 2019).

2.2.10 Research Gaps

- First, the science of ecosystem services is still a new concept in Kenya and even the few local studies have largely employed benefit transfer technique (Seppelt et al., 2011). The application of such techniques assumes that the flow and flux of ES are similar across different forest types, land cover, environmental gradients, and vegetation attributes (Alamgir et al., 2016; Garcia-Nieto et al., 2013; Muller & Burkhard, 2012). This largely attributed to among others, complexity of valuation techniques, and the limited number of local valuations of ES expertise. Worth to note, literature has shown that stock, flow and flux of ES vary from the landscape, vegetation type, and their respective properties (Baral et al., 2014; Burkhard, de Groot, et al., 2012; de Groot et al., 2012). In this aspect, it becomes necessary to assess and estimate an economic value of ecosystem on a local scale. Otherwise deployment of benefit transfer, though commonly used is likely to underestimate or overestimate value of ES of ecosystem. This study therefore narrowed down to two of water catchment ecosystems (Elgeyo and Nyambene) as a local scale-based study. First, because there were no such studies previously carried in the two ecosystems and because the study employed the most acceptable valuation techniques such as cost based and market pricing.
- Secondly, ecosystem generate a wide range of benefits, however, only a small subset of ecosystem services gets it way into conventional markets, thus "priced" (Pascual, Muradian, Brander, Christie, et al., 2010). Commonly, "use values" have market prices while "non-use values/passive use values" have no market "price" primarily because of their public goods and service nature (Diafas, 2014; Martín-López et al., 2007). This is limiting because it does not provide comprehensive ecological values for a given ecosystem. This could be some of the reason it is becoming hard for conservationist to argue and advocate

sustainable conservation in its entirety. The study therefore employed conventional techniques, such as stated preferences to assign and report a monetary value for passive use values. The application and acceptability of stated preference is because the assignment of monetary values to commodities is contingent on societal preference, choices, and trade-offs.

- Society agrees that ESs are fundamental to human wellbeing and welfare (Bateman et al., 2013; De Groot et al., 2009). Literature has demonstrated the linkages between ESs with the economy and human livelihood program and how human development programs affect the stock and flow of ES in the future (UNEP, 2012). However, though the society depends on the nature for its survival and development, its economic value is invisible in policy and decision-making processes. This study attempts to unravel the "whole" value, albeit with complex socioeconomic and ecological linkages. It aims this at provoking sound decision making in conservation of such critical water catchment ecosystems such the Elgeyo and Nyambene.
- In developing countries such as Kenya, rural livelihoods highly depend on environmental resources, such as forests (Najabat et al., 2020). However, dependency on forests is a function of a wide range of attributes including sociocultural, economic, and environmental characteristics (Babulo et al., 2009; Bhavannarayana et al., 2012a; Bwalya, 2013). These attributes vary across landscapes (Coomes et al., 2004; Córdova et al., 2013). In Kenya, there are few studies focus on community dependency on forest, hence, the link between resource use and socio-cultural attributes are not well understood, (Langat et al., 2016). Limited data on forest dependency pose a challenge in understanding household factors that influence forest resource use and threats (Najabat et al., 2020). The lack of such useful information, coupled with anthropogenic pressures in the country, threatens these critical ecosystems and the livelihoods that depend on these resources for survival (Timko et al., 2010). This study therefore attempts to understand how these attributes of particularly forest community can influence state and flow ES from forested ecosystems. This study will be useful in determining empirically supported ecosystem-based policies and

conservation strategies that are compatible with water catchment ecosystems in the Country.

- Local studies have attempted to assess and estimate benefits drawn from a forested ecosystem in the country. However, they hardly demonstrate how anthropogenic drivers such as land use and land cover change, deforestation, over exploitation of forest resources impact on stock and flow of ES. In many cases impact and subsequent cost of human actions have gone unnoticed (TEEB, 2009). This study therefore attempts to not only account for the benefit acquired from the nature but also express and report in monetary terms the impact and cost of degrading our ecosystems. More so, as the society advances in the incentivisation of ecosystem benefits and its pursuance to incorporate ES in projects and policy appraisals. In any case valuation of ES is a prerequisite for the establishment of a market-based mechanism such as payment of ecosystem services (PES) (Engel et al., 2008; Wunder, 2015).
- Similarly, literature associate land cover and use change to decline in stock and flow of ecosystem benefits, livelihood and economy. Although this may be true, what is has been missing is on estimating the level/ degree and cost of change associated particularly with land cover and use changes. In an era of climate change and increased human population, it is crucial to first understand the trend in land use and cover change, distinguish and quantify their impact on future stock and flow ES. This will be critical in assessing and quantifying future availability, management and subsequent planning of critical resources such water, and forest biomass, among others (Njogu et al., 2010; Yaseen, Fu, et al., 2018; Q. Zhang et al., 2011). For water resources it will be crucial in providing early flood risks warning, and avert disasters thus saving lives and resources (Ghumman et al., 2011; Tayyab et al., 2015). This is associated with the need to estimate and project the consequences of land regime change, population pressure, industrial and agricultural expansion (Xu et al., 2014) and climate change, besides the need for policy direction (Arnold et al., 2015).
- Further, researchers have employed a couple of algorithms to estimate and forecast river flow data over years (Adnan et al., 2017; Aichouri et al., 2015; AlMasudi, 2018; Gjika et al., 2019). The last three decades have seen progress in

the application of modelling schemes, including physically based models, the Box-Jenkins method, statistical and machine learning in assessment and predicting river flows and discharge. The physically based model includes HSPF (Donigian et al., 1995), MIKE-SHE (Zölch et al., 2017), and SWAT (Arnold et al., 1998). However, physically based models have limitations as they require a lot of time and a wide range of datasets (Greco, 2012) besides lack predictability function (Jiang et al., 2023). Lack of or part of such information may derail a decision making if it were to be based on physical based model. The biggest gap with this method is that they are unidirectional, and commonly employed in linear data (Di et al., 2014). Notably, hydrological data are commonly non-linear, non-stationary and to some extent are multiscale and multi-directional(Adnan et al., 2017). In that aspect, it becomes a challenge to estimates and predict with certainty the complex non-stationary, non-linear and multiscale data using such a method (Nazir et al., 2019).

- ❖ Likewise, the technological advancement has brought in the application of machine learning (ML) or artificial intelligence, a new paradigm in estimating and predicting river hydrological flows. They include artificial neural networks (ANN), Adaptive Inference-Based Neural Network, Support Vector Machine (SVM)(Nazir et al., 2019), Extreme Learning Machine (ELM) (Ali et al., 2018; Yaseen, Sulaiman, et al., 2018), among others. The ML though highly credited over the traditional methods because of its robustness (Faruk, 2010; Landeras et al., 2009), it however requires large data for training, over-fitting, and slow convergence speed, among others (Adnan et al., 2017) which may not be available. For instance, the SVR involves complex quadratic programming and time-consuming procedures (Adnan et al., 2017). This study therefore endeavours to employ bi-directional, easy to process, interpret and robust statistical methods that can predict stock and flow ES. Such a model should be robust enough to consider complex physical process and dynamic variability of ecological systems such hydrological and meteorological dynamics.

NB: Though it appears ambitious, valuation of ecosystem services is not a condition in decision making but would rather complement other indicators such as the Indicators for Sustainable Development Goals (SDGs).

CHAPTER THREE

MATERIAL AND METHODS

The section gives an account of the procedure, approach, and techniques employed to perform the study. It entails the description of the study area, study design, sampling design, data processing, and analysis. This is as described here below.

3.1 Study area

The study was conducted in two of the water catchments in Kenya, namely the Elgeyo and Nyambene ecosystems. These are among the most productive and diverse ecosystems in Kenya (KWTA, 2020b, 2020c). The two selected water catchments represent the two distinct landscapes and watershed management regimes in the country, that is commercial exotic forests with lesser restrictions and native forests with restricted access. The Elgeyo is largely an industrial forest with less restriction in terms of access, where the community undertakes some farming. While the Nyambene ecosystem is mainly a native forest preserved for non-consumptive use with a restricted extractive harvest, both as described below.

3.1.1 Elgeyo ecosystem

Elgeyo ecosystem is part of the Rift valley and Lake Victoria Drainage basin largely situated in Elgeyo Marakwet County and a small section stretches to Uasin Gishu County. Geographically positioned between 35° 20" and 35° 45" East Longitude and 0° 10" and 0° 20" North latitude. In the year 2019, the Elgeyo water catchment ecosystem covered about 108,367 ha comprising a gazetted forest (25,400 ha) and the buffer zone (82,967ha). The State forest includes Kessup, Kaptagat, Sabor, Penon, and Kipkabus forest blocks (Figure 3.1). Commercial exotic forest covers forty percent of the state forest, while native forest covers approximately 38%, and open grassland and bushland cover 22%. The bigger section of the ecosystem falls in high altitude agroecological zone with a moderately cool climate. While a small section falls within the great rift valley escarpment with a moderately dry and hot climate. It is also part of the Cherangany ecosystem and the rainfall regime is binomial, with the highest recorded between March and May, and short rainfall received in August and October. Average rainfall varies between 700 mm in drier sections to 1400 mm in the highlands. Temperatures range between 11.2^0c to 26.3^0c though could record as 33^0c at parts bordering the Rimoi area (KALRO, 2020). There are significant climate differences because of altitude variations, with Kerio Valley at 1000 m and the highlands over 3350 m above sea level. The slope varies from 2% in the valleys to 37% in the hills (County Government of Elgeyo Marakwet, 2018; The Republic of Kenya, 1980).

The ecosystem is a biodiversity hotspot in the country which supports ecological functions, the wood industry and generates a wide range of ES such as timber, fuelwood fresh water, fodder for livestock, food production, biological diversity, maintenance of soil fertility, and climate regulation among others (KWTA, 2020b).

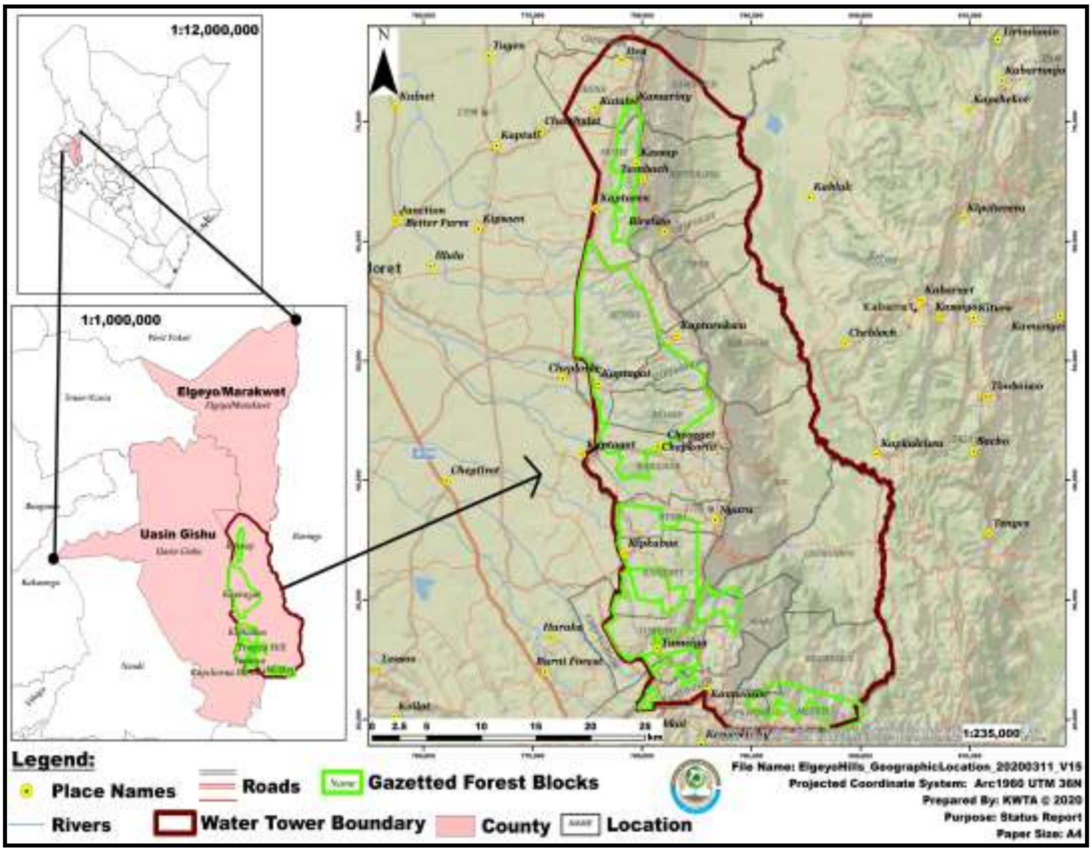

Figure 3.1: Map of Elgeyo Ecosystem Alongside Kenya's and Elgeyo Marakwet County (Source: KWTA GIS Database)

3.1.2 Nyambene Hills Ecosystem

Nyambene Hills is one of the gazetted water catchment ecosystems within the Tana and Ewaso Nyiro River drainage basin and covers an area of 30,313 ha. This comprises the gazetted state forest (5,427ha) and the five-kilometre buffer zone (24,886ha). Cropland dominates the ecosystem at 39%, followed by wooded grassland and dense forest at approximately 25% and 24% of the total cover, respectively. The ecosystem is located in the northern part of Meru County and traverses Igembe South, Igembe Central, Tigania East, and Tigania Central Sub-Counties. It lies between latitudes 0º 17'N and 0º 8'N and longitudes 37º 48E and 37 º 52' E (Figure 3.2). Nyambene's climate is primarily influenced by altitude and mountainous relief. The temperatures range from cool-humid to hot and dry, recording an annual mean of 24.7 ^{0}C at low altitudes on the Eastern side, to 13.7^{0}C at the high altitudes on the Western slopes of the Nyambene ranges. The rainfall pattern is bimodal with an annual average ranging from 1250 mm to 2514 mm on the Eastern and southern slopes to 380mm to 1000 mm on the leeward side.

According to KWTA, (2020c), over 200 springs and a significant number of streams and rivers are found in the ecosystem. These include networks of rivers, such as Ura, Tananthu, Bwathonaro, and Thangatha, which originate from the eastern divide of the ecosystem and drain into the Tana drainage basin. Likewise, River Likiundi and Liliaba are on the western divide and drain to Ewaso Nyiro. The watershed is critical for the provision of water for domestic, irrigation, and industrial use, benefiting both the bordering communities and those further downstream. For instance, communities with the support of the Kenyan government have benefited from the four dams constructed along the Ura River. These dams serve parts of the Igembe South, Central, and North Igembe Sub-county. Previous studies have recorded over thirty plant species belonging to 16 genera in 10 families with the most common species in state forests including *Acacia spp., Cordia abyssinica, Vitex keniensis, Cordia africana, Prunus africana, Trichelea emetica, Grevillea robusta,* and *Commiphora siniensis while Eucalyptus* species, *Cupressus lucitanica* are common in private farms (Lengkeek, 2003). The ecosystem also hosts diverse fauna, particularly primates including (*Papio anubis, Colobus guereza, and Cercopithecus albogularis* among others (KWTA, 2020c).

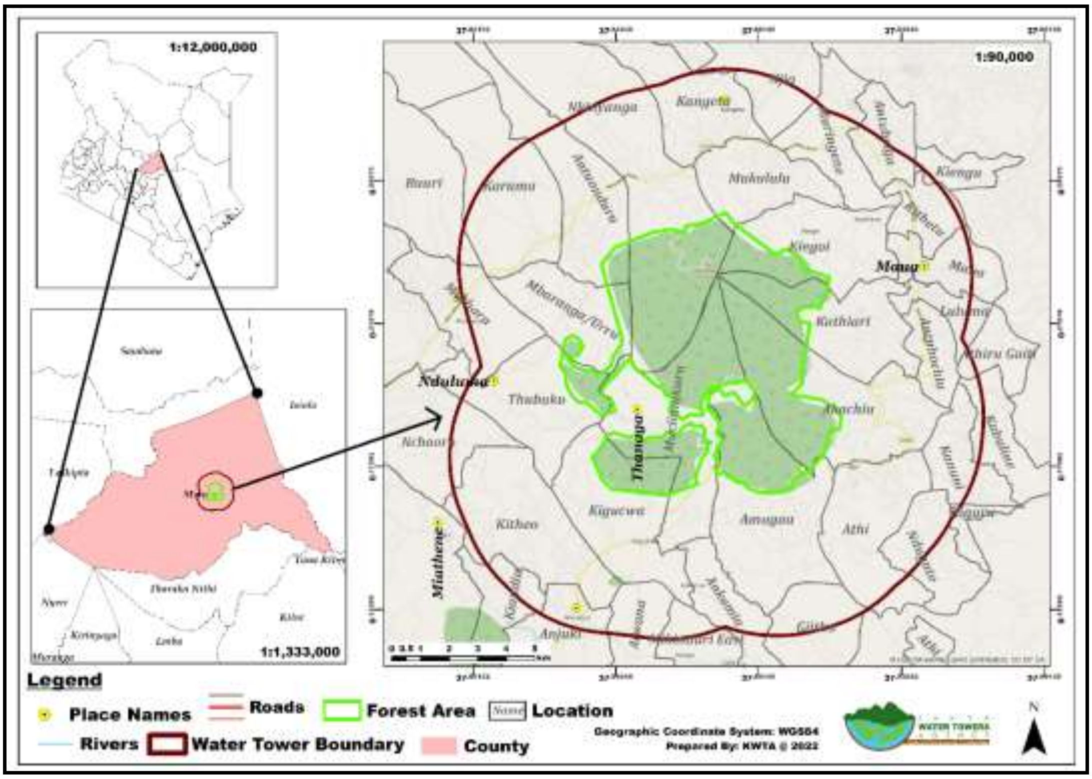

Figure 3.2: Map of Nyambene Water Tower Alongside Kenya's and Meru County

(Source: KWTA GIS Database, 2020).

Gazetted as a state forest in 1969 under the then Forest Department, currently Kenya Forest Service, covering approximately 6,878.40Ha including the Nyayo Tea zone belt that stramong otherishes approximately 120Ha. The state forest comprises four blocks, including Nyambene, Kirimandingiri, Keiga, and Thuuri. The Nyambene block is the largest, while the Thuuri block is the smallest of the blocks. Native forest dominates the ecosystems with less than one percent of the industrial forest. The ecosystem hosts about 35 plant species belonging to 16 genera in 10 families found with about 10 of the species encountered being endemic. Besides the diverse flora, the ecosystem host unique fauna, including primate species such as *Papio anubis, Colobus guereza*, and *Cercopithecus albogularis* (KWTA, 2020c). It is also a haven of diverse avifauna, some of which are endemic and migratory bird species. From the southern section, the elongated ballast rock range dominates the topography of the ecosystem. The highest peak stands at an elevation of 2,528m above sea level. The slopes are very steep and rocky, especially to the eastern side, but the crests are much lower, about 1000m above sea level. Volcanic activity influences the geology of the ecosystem. This comprises basic and intermediate rocks, including phonolites, trachytes, basalts, kenytes syenites, and pyroclastic rocks (KWTA, 2020c).

3.2 Study Population

The study targeted the population living in the two ecosystems spread across the respective administrative units and boundaries. The two counties traversing the Elgeyo ecosystem have a population of about 1.7 million (404,804 households) (KNBS, 2019b). However, of interest to the study was a population of about 405,111 (99,119 HH) from four sub-counties, Keiyo South, Keiyo North, Ainabukoi, and Moiben, perceived to directly interact with the Elgeyo Ecosystem. While in the Nyambene ecosystem, the study targeted a population of about 691,298 (173,743 HH) spread over four sub-counties of Meru County, namely Igembe South, Igembe Central, Tigania East, and Tigania Central as shown (Appendix VII).

3.3 Sample Size Determination

The sample population was drawn 99,119 households (Elgeyo), and 173,743 households (Nyambene) (KNBS, 2019b). This was determined using the Mugenda and Mugenda 1999 formula (3.1).

$$N_{HH}=\frac{Z^2Pq}{d^2} \tag{3.1}$$

where N is the sample of the household, Z is the critical score standard deviation at a 95% confidence level, P is the proportionate standard deviation of the household, q=1-P, while d is the standard error with a *p-value of 0.5.*

The total sample was 384 per site since the household population was over 10,000. The sample was proportionately distributed amongst the lowest enumeration area (Locations) .

3.4 Sampling Design

The study employed mixed approaches and procedures based on the study objective namely, research design on assessment of the forest exploitation levels, perception of ES benefits, socio-economic attributes, and forest dependency. Then, the determination of the total economic value of ecosystem services. Assessment of the impact of land use change on the state and flow of ES using biomass and river flow dynamics. Finally, modelling of the impact of forest cover, change on stock, and flow of ES using forest biomass and river flow dynamics. This is as presented here below.

3.4.1 Forest Exploitation, Perception of ES, Benefits, Socioeconomic, and Forest Dependency

The study adopted a cross-sectional design based on the site-specific assessment as a general research principle (Deaton, 1997). It opted for this design because of its robustness to generate adequate data and describe the status of a product over the years. (Mugenda & Mugenda, 2003). The data sourcing was through consultation, interviews, and field surveys supported by GIS and remote sensing and complimented by secondary data sources. The study conducted the sociocultural and economic survey between November 2020 and July 2021, targeting forest-adjacent communities. This included households, state and non-state representatives (key informants), and focus group discussion.

The study employed a stratified sampling design to delineate and designate enumeration areas within the local administrative unit (location). The enumeration area's sample was determined by multiplying the total sample size with the ratio of the respective number of households to total households (KNBS, 2019b). Enumeration areas (locations) were used and the sampling procedure involved listing the number of households within them. Relative population proportion was used to calculate the number of households per area. The respondent households selection from the list provided by the local administrators employed stratified and simple random approach. The selection of the respective key informant was purposive and based on familiarity with the ecosystem, people, and in-depth understanding of community and forest resources related conflicts and history relevant to the study.

Although the interviews were directed toward household heads, alternate household members would occasionally take their place. This is besides family group discussions aimed at validating and supplementing the household head's information. The study administered both open and closed semi-structured questionnaires triangulated to generate information on forest extraction at the household level. This was validated through the key informants' checklist and focus groups (Achola & Bless, 2006; Kothari, 2004). The information generated included quantities of products collected per visit, time spent, number of household members involved, frequency, and product unit price. This was besides information on household socio-economic and household demographics information (gender, education level, and primary occupation), among others. Quantification of direct use values was based on the household's reports, including quantities of products for domestic use and sale weekly.

Depending on the product in question, the information provided was based on the shortest recall period to reduce memory lapse errors. The inconsistencies on a unit of measure was corrected using a standardised measure, such as kilograms. This includes weighing the products reported in a localised unit of measure such as head load to get the actual weight. The sampling tools pre-test and subsequent piloting were carried In Nzaui water tower Makueni in 2019. The respondent strategic bias (such as concealing or misinformation) corrected through a deliberate effort to define the study's purpose to enhance acquiescence (Kreitchmann et al., 2019). Equally, the involvement of the local leaders as part of the research crew and local research assistants built respondent confidence. The research assistants were trained on questionnaire content, study objectives, interrogation techniques, and the handling of respondents. To reduce respondent suspicion, the crew were accompanied by local village elder as a guide, and each research assistant was deployed in a familiar locality.

The study selected at least three focused groups from each of the two ecosystems, with a composition of between 8 and 10 of all genders. Focused group discussions (FGD) were selected based on their key role in conservation, familiarity, and willingness to contribute to the study. The first role of the key informant was to identify and rank ecosystem services drawn from the two water catchment ecosystems. Secondly, vaidate ES services checklist, besides the discussion of the status, trends, challenges, and priority interventions to be undertaken. Three key informants were each selected from enumeration areas drawn from state and non-state actors. This included the local administrators, Kenya Forest Service, Water Resources Authority, County Government, water service providers, and forest products vendors, among others. The focus group discussion delved into the interaction between the local community and the two ecosystems (trends on forest covers, community history, livelihoods, threats, challenges, among others.) and ecosystem benefits and beneficiaries. While, it primarily aimed the forest product vendors' interviews at generating information on harvest volume, source, and the relative proportion, number of individuals engaged, sales, and unit price.

Forest Dependency

The study used participatory rural appraisal (PRA) to determine the relative household economic status and wealth ranking (Adams et al., 1997). This has commonly been employed in similar studies (Kalaba et al., 2013; Sapkota & Oden, 2008). The economic cohort was based on total household income defined as annual estimates accrued from income sources such as forest products sales, farm, salary, and remittance, among others (Garekae et al., 2017; Najabat et al., 2020). Forest income in the study includes income generated from wood and non-wood forest products, such as timber, honey, among others. The study estimated this using the annual household quantities multiplied by the relative market prices (Hussain et al., 2019). Livestock income included sums of money from the sale of animals and their products, while farm income includes crop yield sales. The study used forest dependency scores to estimate the proportion of forest income contribution to household income (Gatiso, 2017).

The total household income is the summation of income from the forest, livestock, business, on-farm, remittance, rental and lease, pension, and off-farm income, among others (Garekae et al., 2017; Langat et al., 2016; Najabat et al., 2020) refer to equation (3.2). The study did not factor extraction cost in the computation of the net household and forest income because of variability and distorted information on production cost (Campbell & Luckert, 2002).

$$T_{HH\ income} = \sum_{(i=1)}^{n} Y_Z \tag{3.2}$$

Where $T_{HH\ income}$ shows the total household annual income and Y_Z shows income from source Z

Forest income was a summation of household quantities of all the forest products extracted multiplied by the local unit prices (Hussain et al., 2019) That is aggregate monetary value for FPES used by the forest community as shown (3.3).

$$T_{F\ income} = \sum_{(i=1)}^{n} I_P \tag{3.3}$$

Where $T_{F\ income}$ shows total annual forest income and I_P shows income from forest products P

Household interviews were grouped based on their relative wealth status reported in three cohorts (rich, somewhat poor, and poor). In some instances, the survey questions were explained and where necessary translated in the local dialect where the respondent didn't understand English. Research assistants fully explained the purpose and objectives of the study to the respondent and free prior conscent was considered before the actual interviews. Income levels were clustered on an interval of KES 75,000/ (USD700) annually based on poverty baseline of USD 2 per HH per day. A range of between USD 2 and USD 6 a day was categorised as middle income, while income >USD 6 was considered as well off. Any hosehold recording day income less than USD 2 was considered abject poverty cohort. These categories were based on the forest community context and similar studies undertaken in Kenya (Langat et al., 2016) and may not reflect the world bank poverty indexes.

The study computed the forest dependency score using relative forest income defined as the ratio of household aggregated forest income to total household income (Langat et al., 2016) refer to equation 3.5. The mean relative forest income values were used to categorise forest dependency in two levels, that is low and high dependency, with a cut-off point of 0.5 as commonly adopted forest dependency studies (Jain & Sajjad, 2015).

$$F_D = \frac{T_{F\ income}}{T_{HH\ income}} \tag{3.5}$$

where F_D *is the forest dependency score,* $T_{F\ income}$ *is the total forest income, and* $T_{HH\ income}$ *is the total household income.*

3.4.2 The Economic Valuation of Ecosystem Services

The study assessment adopted a typology and groupings within a human utilitarian principle, TEV framework and focused on final products to eliminate double counting (Boyd & Banzhaf, 2007; Fisher et al., 2009b). The choice of the valaution was primarily based on the available data, and ES typology. Based on the TEV framework, the sociocultural and household extraction fell under direct use values, while the ecological process and functions outputs fell under indirect use and non-use values. Both approaches employed market pricing and cost-based, and preference-based, for the assignment of a monetary unit, as presented below.

3.4.2.1 Direct Use Values

Quantification of direct-use forest products used household own reported quantities through interviews. The information generated included product quantity per trip, household members involved, product collection time, frequency per week per product, and gate unit price. The data generated was used to populate weekly household harvest and extrapolate to annual quantities. As aforementioned, communities don't extract all forest products frequently. Collection and use of forest products such as fencing and building poles, timber, and honey among others for domestic use are seasonal or yearly. The study computed separately this type of product from the weekly products.

Overall, the study employed market pricing for the valuation of the direct use of forest products traded in local markets, using the local gate product prices as commonly employed (Campbell & Luckert, 2002; Godoy et al., 2000; Langat & Cheboiwo, 2010). Estimations are based on annual household quantities and relative mean unit price, less production cost (Godoy et al., 2000), as shown in equation (3.6). However, the study used the cost of surrogate units to value products with an unclear market price, such as fodder, water (domestic and livestock), and game meat, among others (Emerton, 2001; Langat & Cheboiwo, 2010; Mogaka, 2001).

$$T_{NDUV}=\left(Q_{HH}\times P_{FP}\right)-C_P \qquad 3.6$$

where T_{NDUV} *is the total net value for direct use of the respective product;* Q_{HH} *is the annual household quantity;* P_{FP} *is the annual mean forest product price; and* C_P *is the cost of production (cost of collection, transport, and other related transaction). The production cost can be estimated by the time spent in the collection where the production cost is not either distorted or missing.*

3.4.2.1.1 Water for domestic and livestock use

The study calculated the amount of domestic water uptake for each household based on their daily consumption. The livestock water consumption was based on the livestock water demand. According to Peden (2003), water contributes up to 80% of an animal's body weight, and water intake depends upon the size of the animal, feed, and salt ingested, lactation and ambient temperature, and an animal's genetic adaptation to its environment. Water intake levels of livestock range from about 5 litres/TLU in cool wet weather to about 50 litres/TLU in hot dry conditions (Appendix II) (IWMI, 2007; Peden et al., 2003). The provision of water has no clear and/or defined quantification processes and market. In that regard, the study employed the yield and cost of an alternative water source (Bush, 2009; Sjaastad et al., 2003) equated to the value of water provision. Studies commonly use sinking a borehole as an alternative water source as employed in ASAL areas globally, though not the only alternative water source (Bush, 2009). In that regard, the study equated the value of domestic (human and livestock) water to the cost of sinking a borehole. According to Dennis Ayemba, the entire borehole process, including surveys, actual drilling, and installation of the pump, may cost anything from KES 1.2 m to 3 million (Ayemba, 2018). The study applied an average cost of KES 3 million to sink a borehole with an average yield of $3.5m^3$ per hour, as suggested by the Athi Water Services Board (AWSB, 2015).

3.4.2.1.2 Forest Grazing/Browse

The two ecosystems are critical in livestock production and essential for the provision of pastures more so during the dry season. Direct quantification of livestock pasture, particularly for free-range grazing, employed surrogates (hay) as commonly applied forest forage valution (Langat & Cheboiwo, 2010; Langat, 2016). First, the study estimated the dry matter demand and then equate it to a fodder substitute (hay). According to the US National Research Council (2001), the daily dry matter requirement is approximately 3% of the body weight. Notably, livestock weight varies across regions, in sub-Saharan Africa, cattle weigh between 150 to 450 kg, and camels weigh between 250 and 550kg. While goats weigh between 15 and 25 kg and the common Persian blackhead sheep weigh between 15 and 35 kg (Kwai et al., 2020). The study used the mean livestock dry matter requirement and the respective tropical livestock unit (TLU) to estimate the total dry matter requirement, as stated in equation (3.7). The TLU, in this case, equated to 250kg (IWMI, 2007; Jahnke, 1982; Peden et al., 2003). This corresponds to 1.4 TLU for camel, 1.2 TLU for cattle, 0.1 TLU for sheep and goats, 0.2 TLU for pigs, 0.3 TLU for donkeys, and 0.01 TLU for poultry (Rothman-Ostrow et al., 2020). The study extrapolates the daily dry fodder requirements estimates to the annual fodder requirement and equated it to bales of hay.

$$TDM_L = (TLU_{LH} \times DMR_{TLU}) \quad 3.7$$

Where TDM_L is the total dry matter requirement for the livestock; TLU_{LH} is the per-head Tropical Livestock Units, and DMR_{TLU} is the minimum dry matter requirement for one tropical livestock unit (Livestock weight multiplied by 3%).

Finally, the study estimated the forest grazing/browsing value using the unit cost of a surrogate (hay) as stated in the formula below (3.8).

$$V_{FG} = \left(\frac{TDM_L}{\omega_S}\right) \times R_S \times \$_S \quad 3.8$$

where $\mathbf{V_{FG}}$ is the economic value for forest grazing; TDM_L is total dry matter requirements; ω_S is the weight of per unit surrogate; R_{SF} is the surrogate (hay) to forest browse ratio; $\$_S$ is the unit cost of the surrogate (hay bale). The average bale of hay weighs about 25 kg, with an average cost of KES 250. Literature commonly equated the ratio of hay to forest browse to 0.6 (Langat, 2016).

3.4.2.2 Non-Use Values and Cultural Values

The study employed contingent valuation (CV) to determine the monetary value for non-use services (bequest, existence) and cultural (recreational services). The technique adopted the commonly applied guidelines (Hanley et al., 2007; Riera & Signorello, 2013) which include establishing the hypothetical market, description of scenarios and economic instruments, and a range of bids to choose from. The CV questionnaire adopted a single-bound dichotomous choice (SBDC) (Fogarty &

Aizaki, 2019) where the respondent had to make one choice either of 'Yes' or 'No' on the 'ill'ngne's 'o pay (W'P).'The 'ue'tionnair' ha' fou' a'pects, i'clu'ing 'n 'nquiry o' th' con'ci'nce of t'e ecosystem and serv'ces. These hypothetical scenarios presented the option of enhanced benefit flow after an intervention vis-à-vis maintenance of the *status quo*. This was followed by a question of whether they agree or otherwise on change in benefit status after an intervention; and lastly choosing from amongst the bids quoted, which range from USD 0 to 200 (Bamwesigye et al., 2020; Petrolia et al., 2014). The study considered every value quoted by the respondents, however, a zero value would be invalid and omitted if quoted as a protest (Eregae et al., 2021). The study also recorded socio-economic and cultural traits (sex, education, income, distance, period of residency, and livestock, among others) to establish their influence on the amount quoted and the willingness to pay.

3.4.2.3 Indirect Use Values

The cohort chiefly generates data on regulatory and support ecosystem services, including water regulation and water purification, climate regulation, soil erosion control, and soil nutrient regulation, among others. Quantification of these services is complex largely attributed to the paucity of data on different indicators. This necessitates the use of indirect techniques, such as replacement, avoided cost, and net productivity change to determine their monetary value. The study employed field surveys, geographical information systems (GIS), and remote sensing to collate this ES. Field-based and GIS data were supplemented by secondary information sources from state and non-state databases. The Ministry of Agriculture and Kenya Bureau of Statistics provided agricultural productivity (crop and livestock) data (MoALF, 2014, 2016, 2018). Equally, the study sourced the hydrological data, water yields from rivers, and annual water use and borehole characteristics from state institutions. This includes the Water Resources Authority (WRA), Water Service Boards (WSB), and Water Service Providers (WSP) (Langat et al., 2020). While it sourced the sediment yield and soil loss data from published sources (Gizaw et al., 2021; Okelo et al., 2009).

The assessment of indirect use values involved biophysical quantification and the assignment of the monetary unit using non-market valuation techniques. The valuation technique used was based on the study scope, data requirement and availability, resources at disposal, subject, and expertise (Baral et al., 2017; Burkhard et al., 2010; Burkhard, Kroll, et al., 2012; Häyhä et al., 2015; Paudyal et al., 2015). The biophysical includes the ecological dimension estimated per unit area, percentages, and scores reported annually. While the monetary unit represents either the direct market price or the surrogate monetary unit price of the ecosystem services quoted (Mengist et al., 2020). The assessment began with land cover and land use, profiling, and quantification of RES using GIS and remote sensing, and field surveys. This information was complemented with secondary data sourced from state and non-state actors' databases. It then assigned the actual or shadow prices for the products and estimating the aggregate values.

3.4.2.3.1 Land Cover Land Use (LCLU) Classification

The study employed Geographic Information System (GIS) and Remote Sensing (RS) techniques with 30 m spatial resolution to generate land cover and land use (LCLU) data for the two ecosystems. LCLU assessment started with imagery generation, image processing, classification using a random forest classifier, and generation of respective classified maps (LC1990, LC2000, LC2010, LC2020). The study downloaded the four Landsat satellite imageries for path/row from three types of sensors from the United States Geological Survey (USGS) website, https://earthexplorer.usgs.gov/ (Table 3.1). The study map images were processed using ArcGIS 10.7 and R Studio 1.4.1106 and ENVI 5.3, generated during the dry season of the year. This was between January and March, to ensure cloud-free and enhanced image visuals.

Table 3.1: Summary characteristics of the path of the Landsat image

Site Name	Image Title	Path/Row	No. of Bands	Spatial Resolution	Acquisition date
Elgeyo/Nyambene	L5 TM	170060	7	30 m	1985/01/16
Elgeyo/Nyambene	L7 ETM+	170060	8	30 m	2000/03/06
Elgeyo/Nyambene	L7 ETM+	170060	8	30 m	2010/01/29
Elgeyo/Nyambene	L8 OLI/TIRS	170060	11	30 m	2021/01/15

The study projected the imageries generated to the Universal Transverse Mercator (UTM) coordinate system, Datum Arc1960, zone 36 North, and corrected for geometric errors from the sources using Ground Control Points sourced from a topographical map at a scale of 1:50,000. The other three earlier version Landsat images (1985 TM, 2001 TM, and 2010 ETM+) and referenced by performing the image-to-image registration method. Referencing employed the newest version sensor that is the corrected Landsat operational land imager/ thermal infrared sensor (Landsat 8 OLI/TIRS). The 2022 image was used as a reference image.

The study employed the first-degree polynomial method and nearest neighbour resampling technique to perform the geometrical rectification and image registration. Landsat Gap-fill extension in ENVI 5.3 was used to correct the line gap appearance from the earlier version Enhanced Thermal Mapper (ETM+) Landsat7 sensor. The study adopted the IPCC scheme II classification (Domke et al., 2019) that considers ten (10) classes, namely dense forest, moderate forest, open forest, wooded grassland, open grassland, perennial cropland, annual cropland, wetland, open water, and bare land. Five kilometer buffer zone was included in the study area.

Ground truthing was conducted using the sample points created from Sentinel land cover maps of the two catchment ecosystems using GPS. Navigation to particular sample locations is accomplished using the MapInr application, allowing field point verification by determining the type of land cover class present on the ground within the forest and its environs. This entails assessing the major land cover type at that point and noting the Scheme I and Scheme II category of the class. Field manouevre was supported by local guides who also shared indigenous knowledge of the ecosystems over the year.

Applied pixel-based supervised classification using a Random Forest classifier in the R-Studio programming interface to limit over-fitting, reduce errors, enhance processing speed, easier validation of data, and generation of the confusion matrix. The process began with polygon training site delineation, land cover coding, and improvement of the image features using true and false colour composite. The study performed validation of the predefined land cover Landsat images training site through the field observation from the 100 assessment points per ecosystem, Google Earth Imagery, and land cover historical data generated through interviews with the adjacent community. With an accuracy of 0.8 based on the class confusion matrix, the study deployed a random forest classifier to create a spectral signature and classification of all the pixels in the image generated. Finally, the study employed an imagery filter to smoothen the classification results by removing 'salt' and 'pepper' noise from the classified maps. The final land cover maps were used to generate and analyse the LCLU class area size (ha) using the 'tabulae' area algorithm in ArcGIS version 10.7 that intersects the images in the respective study area. The resultant canopy classes are based on the forest definition described as being 15-40% tree cover (Open forest), 40-65% tree cover (Moderate), and above 65% (Dense forest) (MoE & F, 2019b).

3.4.2.3.2 Watershed protection

The quantification of the water regulation function adopted the water storage method using the formula shown (3.9) as the most commonly adopted method (Langat et al., 2020; Langat, 2016; Xi, 2009). The study determined the amount of water preserved using the annual precipitation, less evaporation, and runoff, multiplied by relative land cover coefficients. While the value of watershed protection was based on the unit cost of a surrogate (water dams) based on the replacement cost principle (Wu et al., 2010).

$$V_{WFR}=\sum_{i=1}^{N} A_{LC} \times P_C \times RR_{Coef.} \times C_{Prox.} \quad 3.9$$

where V_{WFR} Represents the economic value for water flow regulation; A_{LC} represents the area (ha) of the land cover; P_C represents the average annual precipitation received by the ecosystem; RR_{Coef}. Runoff reduction coefficient of the respective land cover (estimated by the precipitation runoff coefficient of the respective land cover/land use subtracted from runoff coefficient of bare land); C_{proxy} represents the unit cost of metric cubic of water of the surrogate reservoir.

The precipitation runoff coefficient varies from an ecosystem, landscape, climate, and region where, for instance, in Africa, it ranges from 0.01 to 1 (Blume et al., 2007; Goel, 2011; Karamage et al., 2018; Kauffman et al., 2007). However, since the study was site specific, it opted to use explicit runoff coefficients of 0.6 and 0.4 for the Elgeyo and the Nyambene ecosystems respectively equated to the respective forest cover proportions, which stood at approximately 40% and 60% respectively. This assumes that, for instance, 100% forest cover equated to a 0.01 runoff coefficient, while 10% equated to a 0.9 runoff coefficient (Table 3.2). The study adopted the respective land cover and land use runoff reduction coefficients from the literature (Kateb et al., 2013; Okelo et al., 2008, 2009). Since the ecosystem service is non-marketed, the study applied a surrogate in this case sand dam. The unit storage cost of sand dams ranges from 1 to about 5 USD per m^3 with a minimum life span of 50 years and storage of at least 2-4 meters in height and 20 meters (length) (Eytan & Spuhler, 2020; The Ministry of Water and Irrigation & World Bank Kenya, 2005; World Bank Group, 2011). Clay-constructed subsurface sand dams in Kenya with a capacity of 425 m^3 to 1340 m^3 cost between about USD 900 and 1600 (Nissen-Petersen, 2006). This translates between USD 1.2 and 2.10 per metre cubic, although the study opted to use the maximum unit value as it takes care of contingency expenses associated with dam maintenance.

Table 3.2: The summary of precipitation runoff and soil loss coefficients

Land Cover	Runoff Coefficients	Reference Soil Loss (Ton/ha)	RUSLE C-Values
Dense Forest	0.01	1	0.01
Dense Exotic Forest	0.01	1	0.05
Wooded Grassland	0.43	5	0.05
Bushland/Scrubland	0.54	2	0.2
Crop Land	0.54	42	0.15
Perennial Cropland	0.54	8	0.17
Vegetated Wetland	0.01	0	0
Other lands, such as bare, settlement	0.90	70	0.6
Open water	0.01	0	0
Fallow Land	0.90	5	0.6

3.4.2.3.3 Water quality regulation

Studies commonly assess water quality regulation based on the avoided cost principle. This is with an assumption that the degradation of the forest ecosystem would impair this function and thus deteriorate water quality. The destruction of the forest ecosystem would require the establishment of a treatment plant to replace the forest ecosystem's water purification function. The existence of the forest ecosystem would, however, save society the cost of establishing an artificial treatment plant. In that regard, the study would use the cost of the establishment of an artificial treatment plant as a surrogate for estimating the relative unit value of a forest water purification function. Computation can either be through the amount of water used by the society including but not limited to domestic, industrial, and agricultural among other uses, or the amount of water preserved by the ecosystem multiplied by the surrogate unit cost (Jahanifar et al., 2017) as shown in the function (3.10). This assumes the degradation of the forest ecosystem and demands the establishment of a treatment plant to replace the forest ecosystem's water purification function. The study adopted the unit water treatment cost of USD0.3/m^3 (Fuente et al., 2015) as a proxy for the ecosystem water purification function.

$$V_{WQ} = Q_{WC} \times \rho \qquad 3.10$$

whereby V_{WQ} represents the economic value for ecosystem water quality regulation; Q_{WC} is the quantity of water preserved and purified by the ecosystem, also represented by the total household consumption; ρ represents the unit cost of the surrogate water treatment mechanism.

3.4.2.3.4 Soil conservation and Erosion control

Literature commonly quantifies forest soil conservation function using the relative soil loss multiplied by the erosion impact mitigation unit cost. The soil erosion impact mitigation strategies include dredging the water reservoirs, and rehabilitation of degraded areas, among others. As a replacement for the forest soil conservation function, the study selected the dredging strategy. The strategy assumes that soil loss may result in some of it being deposited as sediments in artificial water reservoirs, requiring dredging as a remedy. In that regard, using the sedimentation mitigation unit cost as a proxy, the study computed the value of soil conservation on a mitigation cost principle as shown in equation (3.11). Studies use this approach to estimate the value of both soil and soil fertility protection functions. It equated this to the cost of the alternative goods and services that are likely to be affected in a degraded ecosystem (Bishop, 1999; Nahuelhual et al., 2007).

$$V_{SC} = \sum LC_A \times SE_{RC} \times C_{Proxy} \qquad 3.11$$

whereby V_{SC} represents the economic value for forest soil conservation; LC_A is the respective land cover area (ha); SE_{RC} is soil erosion reduction coefficient based per land cover soil erosion coefficients (Hurni, 1988; Tessema et al., 2020); C_{proxy} Unit Cost of the Proxy estimated at KES 351 (USD 3.34) per ton of sediment (Adeogun et al., 2016).

3.4.2.3.5 Soil Nutrient Conservation

The estimation of soil fertility can be determined in two aspects, that is *in situ* and *ex-situ* effects (Xue & Tisdalle, 2001). *In situ* effects include the reduction of onsite soil minerals (nitrogen (N), phosphorus (P), potassium (K), and organic matter among other minerals). While the *ex-situ* includes indirect effects such as nutrient loss, such as eutrophication in water bodies (Nahuelhual et al., 2007). In that regard, using the on-site soil nutrient soil loss preservation, the study estimates this function by the amount of soil conserved across the landscapes with the relative nutrients content based on field soil sampling and laboratory analysis.

Estimating the economic value for forest soil fertility conservation was based on the replacement cost-based principle. The study assumes the loss of forest soil would lead to the loss of soil nutrients that are critical to plant growth and demands the use of alternative artificial fertilisers to replace the requisite soil nutrients. The study used commercial fertiliser in this case as a surrogate for the soil nutrients' loss. Using the market unit prices for commercial fertilisers multiplied with the relative land cover soil loss coefficients as shown in equation (3.12) the study computed a shadow value for forest soil nutrient conservation based on a replacement cost principle (Gizaw et al., 2021).

$$EV_{SNC} = \sum CS_{LC} \times SN_{LC} \times \delta_{SNF} \times P_{CF} \qquad 3.12$$

where EV_{SNC} is the economic value of soil protection; CS_{LC} is soil conserved (kg/ha) of the respective land cover; SN_{LC} is the soil nutrient content (%) (N, P, K) in the forest soil; and δ_{CF} is the ratio of commercial fertilisers (1/51%, NPK-17-17-17); P_{CF} is the unit price of the commercial fertilisers (KES 60/kg).

3.4.2.3.6 Tree carbon quantification

The study employed double-stratified two-phase systematic cluster sampling. First overlays of 2 x 2 km generated grids, followed by stratification into rectangular clusters of six sample plots 250m apart (Figure 3.3). The selection of the rectangular cluster design was based on the location of the study site falling within stratum 2 (Figure 3.4) of the national agroecological zoning. These include Stratum one (arid and semi-arid-ASAL), Stratum 2 (Highland/ arable region), stratum 3 (coastal area), and Stratum 4 (mangrove ecosystem). (KFS, 2016).

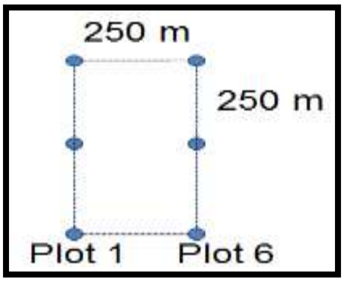

Figure 3.3: Second Phase

(Sample Plot Sampling) Design

(Adopted KFS, 2016)

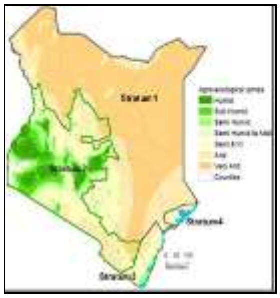

Figure 3.4: National forest inventory stratification, Kenya

(Sourced: (KFS, 2016))

The first step was to locate the plot using a high-precision GPS gadget, demarcating the nested concentric plot and marking. The actual selection of the first plot was random, though based on the proximity from the point of access to the systematically selected point. The study adopted a nested concentric sample plot design based on reports of enhanced plot data accuracy. This was important when sampling mature forest ecosystems with large trees, consequently saving time (KFS, 2016). The outer radius of the plot was 15 m used to record and measure trees with DBH ≥20 cm, the inner 10 m radius circles to measure trees with DBH ≥10<20 cm, and the radius 5 m circle on the hand was used to measure trees with DBH ≥5<10 cm while a radius of 2 m for the tree with DBH <5 cm That is measure seedlings. Established regeneration subplot at radius 10m circles, a small white circle on north-south of 10 radius circle and the optional regeneration subplots small steric notation white circles positioned east-west on radius 10m circle (figure 3.5) explained and established in the regeneration circle (figure 3.6).

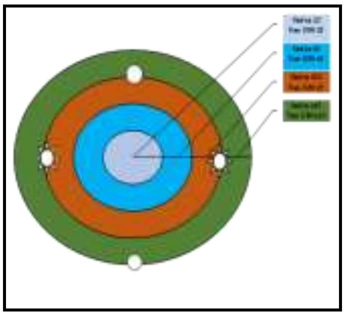

Figure 3.5: Nested concentric sampling plot

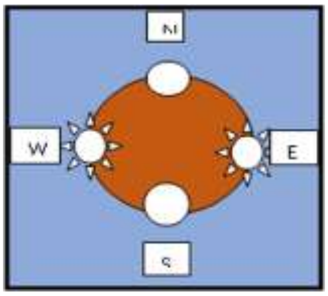

Figure 3.6: Regeneration sampling plots at a 10m radius

(Adapted from the KFS, 2016)

The study collected samples (plant and soil) inside a sample plot while recording area attributes either within the plot or from the surrounding area through observation. The information recorded included land use, selective logging, livestock grazing, firewood harvesting, vegetation type, soil, and forest products, and services. This is besides information on shrubs, regeneration, dead wood, stumps, and other disturbance. The tree dimension recorded were species, tree height and the breast height diameter. Every 5th tree in each selected plot was consisdered as a sample tree where more variables (crown diameter, lower and upper canopy). The study enumerated all the trees encountered in the sampling plot for the computation of density estimates.

The diameter at breast height (DBH) was estimated DBH meter, while the tree height was determined using the upper canopy/height of each tree using a clinometer. The study took the angle between the treetop and the eye view at the breast height angle into consideration for tree height measurement and calculated tree height using the formula (3.13) (Chavan & Rasal, 2010).

$$H = h + (\tan\alpha \times b) \tag{3.13}$$

Where H is the tree height, α angle between the tree and eye view, b is the distance in meters between the tree and observer, and (h) was the height of the horizontal plane of the clinometer equivalent to the observer's eye level height.

Quantification of tree biomass would primarily employ the destructive/harvest method, which is more accurate (Phuong et al., 2012; Vashum & Jayakumar, 2012). However, because of the restriction and the nature of the targeted ecosystems, the study adopted a generalised equation, popularly known as the improved pan-tropical mixed-species model, as shown in equation (3.14) to estimate the aboveground biomass (AGB). The tree biomass assessment targeted two major carbon pools (stem and root biomass) for any tree with DBH≥ 5 cm.

$$AGB = 0.0673 \times (\rho D^2 H)^{0.976} \quad 3.14$$

where AGB is the above-ground weight of the tree (Kg), ρ wood density D is the diameter at breast height in cm, H is the tree height, while α and β are the model coefficients.

The total tree biomass was determined by a factor of 1.25 of the AGB (Chavan & Rasal, 2010) suggesting that the root biomass is approximately 25% of the AGB. The carbon concentration of commonly approximated to be 47% of dry weight (Aalde et al., 2006; Domke et al., 2019). This assumes carbon content is approximately 47% of the total biomass. Wood densities range from 0.276 to 0.551 for the softwood category and 0.6 to >1.1 for the hardwood category according to the wood density database (Zanne et al., 2009). Wood-specific gravity is a significant predictor of AGB, especially when a study considers a broad range of vegetation types (Chave et al., 2014).

3.4.2.3.7 Soil Carbon Assessment

Soil samples were collected from the north point at a radius of 15m of each sampling plot at two profile points (0-30cm and 30-60cm) using a 7.6 cm diameter auger for SOC measurements. Some sampling plots, however, recorded depth restriction, thus the auger penetrates more so the second profile (30-60). The study collected bulk-density samples using conventional core rings and roughed with aluminum foil for safe transportation to the laboratory. A mixture of soil samples from each depth profile was mixed to form a sample for analysis. The organic matter (OM) content was determined using the loss-on-ignition (LOI) method while estimating the organic carbon (OC) employed a ratio of 1:0.58 (SOM: SOC). Soil samples were oven-dried, crushed using mortar and pestle to homogenise, and then sieved using a 2mm mesh to remove debris and stones which were weighed separately. Prior processing was aimed at preserving the soil aggregates while removing larger organic debris before laboratory analysis. The processed soil was then subjected to a dry combustion proceduhttps://www.teamviewer.com/en/download/windows/re as is required for carbon analysis to eliminate any remnant moisture. Placed the two samples weighing 10 grams each were placed in a pre-weighed crucible and then combusted at 550°C for at least 8 hours and then cooled before recording their respective weight. The differences in the mass of weights of the soil before and after heating represented the moisture and organic matter content, while the residue represented the ash. The study derived soil organic carbon (SOC) using the equation shown (3.15) while computing the per unit area carbon using SOC multiplied by respective soil coefficients as shown in equation (3.16)

$$\mathrm{SOC} = \frac{(\mathrm{SSW\text{-}CRW})}{\mathrm{SSW}} \times 0.58 \qquad 3.15$$

where SOC is soil organic carbon (%); SSW is the initial soil sample weight; CRW is soil combustion residue weight.

$$\mathrm{TOC}_{(\mathrm{Mg_C\,Ha^{-}})} = \big((\rho) \times (\mathrm{D}) \times (\mathrm{SOC})\big) \times 100 \qquad 3.16$$

where TOC is total organic carbon; ρ is bulk density; D is soil profile depth and SOC is soil organic carbon.

The bulk density was determined by first weighing raw soil samples, air-dried at approximately 40°C for 48 h, with an aliquot of each sample picked after weighing the air-dried samples. Samples were further oven dried at 105°C for twenty-four hours with their weights recorded. The study recorded three weights for each sample (That is total soil weight, the weight of the aliquot before oven drying at 105°C, and weight after oven drying at 105°C) were then used to compute the respective soil bulk density.

3.4.2.3.8 Carbon Valuation

Valuation of carbon sequestration commonly uses market pricing, although some literature argues that the technique doesn't reflect the true value. A couple of studies have recommended the use of other techniques, such as climate change damage function

(Feraro et al., 2011). That notwithstanding, and because of the complexity (Pearce, 2001) of the climate change damage formula, the study opted to use market pricing using the formula below (3.17).

$$V_{FCR} = \sum_{n=1}^{\infty} A_{LC} \times Q_C \times \varepsilon_C \qquad 3.17$$

where V_{FCR} is the economic value for climate regulation, A_{LC} area (ha) of the respective land cover, Q_C is the quantity of carbon dioxide sequestered per unit area by the respective land cover, while $\mathcal{E}_C$ represents the average global carbon market price per unit carbon.

The global compliance markets currently range from less than USD1/ t CO_2e up to USD 30/t CO_2e (AU\$1-\$29/ t CO_2e). While the voluntary market prices range between USD 1/ t CO_2e to USD 5/ t CO_2e or (AU\$1-\$6/t CO_2e) (World Bank Group, 2020). The study, however, opted to use USD 5 per ton of CO_2 as the prevailing price commonly costs carbon transacted in Kenya.

3.4.2.3.9 Forest description

The study described the forest structure as relative frequency, relative density, relative dominance, and species importance values. These include estimating the relative frequency (Rf) using the number of plots in which a species occurs divided by the sum of occurrences of all species in plots; relative density (Rd) using the number of individuals of a species divided by the total number of individuals of all species; relative dominance (RD) using a basal area of a species divided by the sum of basal areas of all species; and importance value, summation of Rf, Rd, and RD.

The Shannon-Weiner diversity index was used to compute species diversity indices (3.18).

$$H' = -\sum_{i=1}^{s} pi \times \ln pi \tag{3.18}$$

Where H' is the diversity index, pi is the proportion or abundance of the i^{th} *species expressed as a proportion of the total abundance, and ln is the natural log. The measure of species distribution was determined using Pielou's species evenness equation as below.*

$$E' = \frac{H'}{\ln S} \tag{3.19}$$

where: E' is the Pielou's species evenness, H' is the Shannon index and ln (S) is the natural log of the No. of species encountered in the study.

$$BA = \pi\left(^{DBH}/_{2}\right)^2 \tag{3.20}$$

where BA is the Basal Area, π is pie (3.14)

3.4.2.10 Ecosystem oxygen generation quantification

The study estimated oxygen generated by the two ecosystems based on the photosynthesis formula (3.21). In that aspect, the atomic mass ratio of one-ton carbon dioxide sequestered to the oxygen released is equivalent to 0.73.

$$6CO_2(264g) + 12H_2O\ (180g) \rightarrow C_6H_{12}O_6(180g) + 6O_2(192) \tag{3.21}$$

The cost of production of Liquified oxygen (LOX) for small-scale plants in Kenya is about USD 2.10 for 6.8 m^3 (equivalent to the volume of one "J" Cylinder). Notably, the cost only includes a production facility, maintenance, labour, and electricity but excludes the cost cylinder (Institute of Transformative Technologies (ITT) & Oxygen Hub, 2021).

3.4.2.3.11 Microclimate influence

The study employed a productivity function to estimate the value of the microclimate ecosystem function. In this aspect, it used the yield difference between the agricultural production of farmers close to the ecosystem and those further away as a surrogate. Whereby, according to Kipkoech et al. (2011), forest edge microclimate contributes between eight and twenty percent of agricultural production. The study, however, adopted the average proportion of 15% of annual crop yields as a surrogate of forest edge microclimate value.

3.4.2.3.12 Pollination Services

The ecosystem pollination function commonly applies the FAO pollination dependency function designed specifically for insect pollination. The pollination function begins with estimating crop yield data from farmers around the targeted ecosystem (Elgeyo and Nyambene). Then, the study determined the respective annual crop market value from the sampled households. Using the respective crop pollination dependency ratio (FAO, 2005) and using the equation (3.22), the study determined the value of the ecosystem pollination function.

$$V_{NP} = \sum_{N=1}^{i} (PDR_C \times AY_C \times MP_C) \qquad 3.22$$

where V_{NP} represents the economic value for the ecosystem's natural pollination function; PDR_C is the pollination dependency ratio of the respective crop; AY_C is the annual yield (kg) of the respective crop, and MP_C is the market price of the respective crop. Sourced the unit price online (KODI, 2021).

3.4.3 Impact of Land Use Change on the Stock and Flow of ES.

3.4.3.1 River flow assessment

The study assessed the water level and discharge for three gauging stations in the two ecosystems. These include Moiben- station 1BA01 (Elgeyo), Ura- station 4F09, and Thananthu- station 4F20 (Nyambene). The river flow secondary data was from 1953 to 2018 (1BAO1 Moiben), 1957 to 1996 (4F09 Ura), and 1965 to 2001 (4F20 Thangatha). The study primarily aimed at assessing the river flow regimes (extreme flows and volume) monitoring period recorded. First, the survey assessed the key river flow dynamics from daily records over time and later assessed the influence of land cover change (tree cover) on flow regimes. Key dimensions included the mean, base, average, and peak flows of discharge and water level. The assessment was aimed at demonstrating river hydrological and geomorphological dynamics using flow curves, including flooding levels, and predicting flood frequency and seasons (Kibet et al., 2019). Notably, natural phenomenon and anthropogenic drivers have influenced the river flows dynamics over the years. The flow duration curve in that aspect shows both the base flow and peak flow that inform the changes in both the climatic condition and ecosystem bio-geophysical condition.

3.4.3.2 Forest Biomass

The assessment of forest biomass across the different land cover over time employed a mean comparison tool statistical package (SPSS). This began by determining the mean forest biomass across the different land cover followed by an analysis of variance. The land cover classification was based on the forest definition described as being 15-40% tree cover (Open forest), 40-65% tree cover (Moderate), and above 65% (Dense forest) (MoE & F, 2019b).

3.4.4 Modelling the impact of forest cover, change on stock, and flow of ES.

3.4.4.1 Forest Biomass

The test of the generalised linear model (GLM) on Elgeyo forest biomass data revealed varied levels of log-likelihood, Akaike information criteria (AIC), and Bayesian information criteria (BIC), as shown in (Table 3.3). The choice of the most appropriate models was based on the log-likelihood, primarily recording a maximum level, while for AIC and BIC, a minimum value was appropriate. Based on the statistical model summary below, Gamma GLM (log link) scored the most appropriate statistical model (3.23) for modelling and predicting the status of the Elgeyo forest biomass and any similar ecosystem. Primarily general Gaussian linear model (GLM) would fit such data because of its power in controlling type 1 errors (false positive conclussion). However, the study's forest biomass primary data violated the linear model assumptions, particularly on normality distribution and independence, residual independence, and homoscedasticity. In that regard, the study opted for gamma, log link generalised linear models (GLM) since it is flexible and powerful in non-normally distributed continuous data (Ng & Cribbie, 2017). Besides the potential to improve the type 1 error through resampling, particularly for significance testing (Warton et al., 2016).

$$\log Y = \alpha_0 + \beta_1 x_1 + \mathcal{E}_1 = Y_i = \exp^{(\alpha_0 + \beta_1 x_1)} + \mathcal{E}_1 \qquad 3.23$$

Table 3.3: Model Summary (Elgeyo forest biomass)

Statistical Model	ll (null)	ll (model)	df	AIC	BIC
Poisson GLM (log link)	.	-560.537	5	1131.074	1140.430
Gaussian GLM (log link)	.	-239.345	5	488.690	498.046
Gamma GLM (log link)	.	-176.460	4	360.921	367.575

N=48 Observations for estimating the parameters (ll-log likelihood, AIC, and BIC)

Similarly, the test on a generalised linear model on the Nyambene forest biomass data depicted varied scores on log-likelihood, AIC, and BIC, as shown (Table 3.4). In that regard, Gaussian GLM (log link) would also be the most appropriate for modelling and predicting the status of forest biomass for such an ecosystem and any other similar one.

Table 3.4: Model Summary (Nyambene forest biomass)

Model Type	ll (null)	ll (model)	df	AIC	BIC
Gaussian GLM (log link)	.	-143.986	4	295.971	301.155
Gamma GLM (log link)	.	-148.707	4	305.413	310.596
Poisson GLM (log link)	.	-393.362	4	794.724	799.908

Note: N= 27 Obs used in calculating BIC (Bayesian information Criteria), AIC (Akaike information Criteria) ll (Log-likelihood)

3.4.4.2 River Flow Dynamics

A multivariate time series model was used to evaluate river flow dynamics, primarily to determine the impact of land cover on river flows in watershed ecosystems. The study selected a vector autoregressive (VAR) model over univariate or bivariate models due to its superior ability to identify the co-movement of multiple series variables, primarily driven by the lagged values principle. The study opted for the model instead of simultaneous multivariate equations, which are primarily used in applied science (Abrigo & Love, 2016) and macroeconomics (Lenza & Primiceri, 2020; Sims, 1980) including panel data (Holtz-Eakin et al., 1988). In macroeconomics, the VAR model is widely used in estimating, summarising, and forecasting aimed at projecting an economic outlook, thus guiding economic policymakers (Stock & Watson, 2001). A vector autoregression model is a linear form of a model which assumes the variable of its own lagged, current and earlier values, explaining the forecasted values (Stock & Watson, 2001). The study employed a similar economic concept using the VAR model to quantify and summarise river flow with the ultimate aim of developing a VAR model for projecting future river flow dynamics. Equally, the study employed statistical tests within VAR, such as Granger-causality tests, variance decomposition, and impulse response functions, to summarise and reveal variables' co-movement.

However, the model is not devoid of limitations, including the assumption that all the explanatory and response variables are stationary, and errors are 'white noise' which may not be the case in most applied science and economic data. For instance, the impulse error can be misleading for persistent parameters (Kilian, 1999; Wright, 2000), and without adequate modification, the model may not appropriately capture the nonlinear relationship, residual variance, and inevitable error fluctuation (Stock & Watson, 2001). The structural bivariate VAR(1) one lag model is shown (3.24 & 3.25) and bivariate VAR(2) two lag model is shown by (3.26 & 3.27). In that aspect, the study accounted for non-linear fluctuation in river flow dynamics, as influenced by natural and anthropogenic drivers' impulses and changes. Further, the study performed the test for impulse response function (IRF) using the VAR model to measure the effect of a shock on an endogenous variable itself or other endogenous variables (Becketti, 2020; Lutkepohl, 2005).

$$V_t=\alpha_1+\beta_{11}V_{t-1}+\beta_{12}Y_{t-1}+\varepsilon_{vt} \qquad 3.24$$

$$Y_t=\alpha_2+\beta_{21}V_{t-1}+\beta_{22}Y_{t-1}+\varepsilon_{yt}$$

$$\begin{bmatrix} V_t \\ Y_t \end{bmatrix} = \begin{bmatrix} \alpha_1 \\ \alpha_2 \end{bmatrix} + \begin{bmatrix} \beta_{11} & \beta_{12} \\ \beta_{21} & \beta_{22} \end{bmatrix} \begin{bmatrix} V_{t-1} \\ Y_{t-1} \end{bmatrix} + \begin{bmatrix} \varepsilon_{vt} \\ \varepsilon_{yt} \end{bmatrix} \qquad 3.25$$

$$V_t=\alpha_1+\sum_{n=1}^{n=2}\beta_{11}V_{t-n}+\sum_{p=1}^{p=2}\beta_{12}Y_{t-p}+\varepsilon_{vt} \qquad 3.26$$

$$Y_t=\alpha_2+\sum_{n=1}^{n=2}\beta_{21}V_{t-n}+\sum_{p=1}^{p=2}\beta_{22}Y_{t-p}+\varepsilon_{yt}$$

$$\begin{bmatrix} V_t \\ Y_t \end{bmatrix} = \begin{bmatrix} \alpha_1 \\ \alpha_2 \end{bmatrix} + \begin{bmatrix} \beta_{11} & \beta_{12} \\ \beta_{21} & \beta_{22} \end{bmatrix} \begin{bmatrix} V_{t-1} \\ Y_{t-1} \end{bmatrix} + \begin{bmatrix} \beta_{11} & \beta_{12} \\ \beta_{21} & \beta_{22} \end{bmatrix} \begin{bmatrix} V_{t-2} \\ Y_{t-2} \end{bmatrix} + \begin{bmatrix} \varepsilon_{vt} \\ \varepsilon_{yt} \end{bmatrix} \tag{3.27}$$

where V_t and Y_t represent the variables (n), α is the constant, while Vt-p represents the lagged endogenous values, β is the model coefficients, and ($\mathcal{E}$) is serially uncorrelated error

3.5 Data processing and analysis

The study recorded the raw data in Excel and used the pivot table in Excel to process and undertake preliminary analysis, including estimating the diversity index. It later exported this to the Statistical Package for Social Sciences (SPSS) version 24 and the Stata software for further processing and analysis. The study subjected the data to normality tests, identified necessary outliers, undertook the requisite transformation, and eliminated the outlier for compliance with the normal distribution assumption. Data transformation applied natural log, log10, and Inverse fractional rank. The normality test output determined either utilisation of the parametric test (Comparison of mean, one-way ANOVA, t-test) or non-parametric test (Friedman's, Kruskal-Wallis, Mann-Whitney, Wilcoxson) for significance testing. The level of significance was determined at $p \leq 0.05$.

Equally, the study utilised descriptive statistics to summarise household demographics, socio-economic parameters, nature, and ecosystem services quantities using a measure of central tendency and dispersion units. Computed the value of forest provisioning ecosystem services (FPES) using the annual household quantity multiplied by the average market prices and fewer production costs (such as transportation costs or fees payable). The product price was determined by the average product sale monetary unit sourced from local and neighbouring urban markets.

The study employed a non-parametric test (Kruskal-Wallis and Mann-Whitney) for significant testing of forest product household extractions. The study attributed widespread values associated with the heterogeneous forest community (Langat et al., 2016). In that aspect, the study employed logistical regression instead of linear regression for predicting forest dependency, since it violated the linear regression assumption. A collinearity test was also performed before the logistical test and, where necessary, eliminated overlapping and higher variable inflating factors (>0.5) and lower tolerance levels (<0.5) from predictor variables. A binary logistical regression model (Hosmer et al., 2013) was used to determine the predictability of the forest dependency score as a function of socioeconomic and cultural attributes refer to equation (3.28).

Forest dependency score regressed with selected explanatory variables, such as locality, gender, age, education, residency length, household size, land size, TLU, distance, Expenditure-Bands, Income Bands, and place of birth as socio-economic surrogates. The study measured the forest dependency score as a dichotomous response which falls within 0 and 1. Where 0 denotes the least dependency while 1 represents high forest dependency (Garekae et al., 2017). The forest dependency was further categorised into levels That is low and high, using a 0.5 score as a cut-off point and categorised values ≤0.5 as lower tier while those ≥0.5 as a high tier dependency (Garekae et al., 2017; Jain & Sajjad, 2015).

$$\log\left(\frac{p}{1-p}\right)=\beta_0+\beta_1 X_1+\ldots..,\beta_k X_k \qquad 3.28$$

Where p represents a probability that Y=1, given X; Y represents the dependent variable (forest dependency level); X_1, X_2…, X_k represent Independent Variables (socio-economic attributes); β_0, β_1…, β_k Parameters of Model

The generalised linear model (GLM) was used to perform both univariate and multiple regression analyses. While it employed vector autoregressive for modelling the impact of land cover and use changes on river flow dynamics.

Hypothesis Testing

- The study employed multivariate logistical regression test for H_{01}. (on sociocultural and economic attribute influence on forest dependency)
- Independent t-test was used for H_{02} (contribution of ES in local and national income).
- Generalised linear regression was applied for H_{03} (LCLU testing) hypothesis
- Granger Causality test was employed for H_{04} (Tree cover change on river flow dynamics).

CHAPTER FOUR

RESULTS AND DISCUSSIONS

This chapter presents the study findings based on the study objectives, including the socio-economic characteristics, level of exploitation of ES, stakeholder perception of the ecosystem's benefits, threats, and dependency; the total economic value of ecosystem goods and services; the economic impact of prevailing land-use regimes and biodiversity on ecosystem functionality; and a mathematical model on the impact of prevailing land use and biodiversity change to stock and flow of ecosystem services for future prognosis. This endeavours to answer the question of how society extracts ecosystem benefits and its influence on the stock and flow of ecosystem services. Overall, total economic framework classification was used to structure the findings, that is broadly as use and non-use values.

4.1 Forest community socio-cultural attributes, perception of ES benefits, threats, and dependency

4.1.1 Household demographics

The study assessed the social-economic traits of communities living within and around the two ecosystems, including age, period lived in the area, and the size of the household. It estimated the mean age at 45.2 ±12.0 and 44.8±13.7 for Elgeyo and the Nyambene ecosystem, respectively. This age bracket is consistent with most forest community surveys, where most of the respondents fall in the mid-age brackets (Garekae et al., 2017; Rotich, 2019). The mean household size was 6.2±2.2 and 5.6±3.0 for the Elgeyo and the Nyambene, respectively. This is significantly above the Kenyan national average household size, estimated at 4 persons per household (KNBS, 2019a). However, the household size, though slightly lower compared with a study in East Mau, was within range (Langat et al., 2016). Households in the Elgeyo ecosystem revealed significant differences ($p<0.05$) in terms of age, period lived, and household size across administrative units, as opposed to Nyambene, which had non-significant differences (Table 4.1). Overall, the age bracket established in both ecosystems suggests an active population cohort, which implies an increase in demand for socioeconomic necessities in the upcoming years.

Table 4.1: Summary Statistics on Household Demographic

HH Traits	Elgeyo Ecosystem Total (N=373)			Sig. (Locality)	Nyambene Ecosystem Total (N=402)			Sig. (Sub-county)
	Valid Percent	Mean	Std. Dev.		Valid Percent	Mean	Std. Dev.	
Age	95.7	45.23	11.99	0.000	73.6	44.75	13.69	0.141
Period Lived	96.2	38.43	17.36	0.000	75.9	41.92	15.10	0.136
Male	99.2	1.91	1.31	0.000	93.8	2.02	1.45	0.611
Female	98.9	1.82	1.32	0.000	92.8	1.87	1.32	0.896
Children	98.7	2.59	1.77	0.000	73.1	2.57	1.50	0.010
Total HH Size	99.7	6.18	2.22	0.003	99.3	5.55	2.96	0.077

The study also categorised responses based on the social trait cohorts, including gender, marital status, household leadership, place of birth, and ethnic group. Most of the respondents were men at approximately 58% and 60% for Elgeyo and Nyambene, respectively. Most of the respondents reported to be originally from the two ecosystems as confirmed by 75% and 82% born within the Elgeyo and Nyambene respectively, thus very few immigrants. This was inconsistent with a study of East Mau that recorded a higher percentage of immigrants (Langat et al., 2016). In the family leadership in Elgeyo, where the dominant ethnic group was Kalenjin, males headed and managed most of the households, as reported by approximately 67% with less than 8% being headed and managed by women. However, in the Nyambene ecosystem dominated by the Ameru ethnic group, the family leadership, though mostly headed by men, females take a lead role in the management of the family affairs as reported by 53% of the households (Table 4.2). The findings on the community around Elgeyo represent an atypical African family leadership structure where women take a passive role in households and societal decision-making (Kawarazuka et al., 2019). The Nyambene community represents a more liberal, inclusive family leadership setup, as advocated by modern society (Aju et al., 2022; Malapit et al., 2019; Musalia, 2018).

Table 4.2: Respondent Social Traits Frequencies

Social Traits	Category	Elgeyo	Nyambene
		Percent (N=373)	Percent (N=402)
Gender	Male	57.9	60.2
	Female	42.1	39.8
Marital Status	Married	86.9	77.9
	Separated	0.3	3.5
	Divorced	0.3	0.8
	Single	8.3	11.2
	Widowed	4.3	6.7
HH Leadership	Male-headed, male-managed	67.3	31.6
	Male-headed, female-managed	24.9	52.8
	Female-headed, female-managed	7.8	15.4
	Child-headed, child-managed	0.0	0.3
Born in The Area	No	24.9	12.6
	Yes	75.1	87.4
Ethnic Group	Gikuyu	0.3	0.8
	Kalenjin	97.8	-
	Kamba	0.3	-
	Kisii	0.3	-
	Luhya	0.3	-
	Meru	0.3	99.2
	Turkana	0.5	-
	Others	0.3	-

4.1.2 Household education and primary income sources

The assessment of the level of education and primary occupation recorded varied frequencies. For instance, the study recorded that a majority of the population had attended up to secondary levels as reported by approximately 76% and 82% for Nyambene ecosystems, respectively. And approximately 20% of the population within the Elgeyo reported post-secondary education, with less than 5% reported had not attended school. However, the population around the Nyambene ecosystem recorded a 10% for the post-secondary level and a 10% not to have attended school. Overall, a majority of the population was literate and able to read and write, as revealed by the ability to comprehend the benefit of conservation when interrogated. Although not reflecting the actual picture of the entire country, it represents parts of the country that have improved access to education and socioeconomic development (KNBS et al., 2020; KNBS & SID, 2013). In the occupation variable, the study records crop production as the primary source of income as reported by approximately 79% and 62% for Elgeyo and Nyambene, respectively. Equally, a significant population is engaged in business enterprise and craft work (Table 4.3). These findings represent a rural African household where on-farm production tops the list of livelihood and incoming generating activity primarily dictated by favourable weather (County Government of Elgeyo Marakwet, 2018; Davis et al., 2017; MoALF, 2014).

Table 4.3: The summaries of education and primary occupation

Parameters	Category	Elgeyo	Nyambene
		Percent (N=373)	**Percent (N=402)**
Level of Education	None	3.2	9.8
	Primary	30.6	54.1
	Secondary	45.6	26.1
	Tertiary College	16.9	6.5
	Undergraduate	2.9	3.0
	Postgraduate	0.8	0.5
Main Occupation	None	0.8	7.8
	Crop farmer	78.6	61.9
	Pastoralist	0.3	0.5
	Business	12.1	10.3
	Salaried employment	6.4	5.0
	Craft workers/Unskilled	0.8	14.0
	Retired Pensioners	1.1%	0.5

4.1.3 Land and Land-uses

The assessment of households' land and its use recorded a significant difference across administrative units. In Elgeyo, for instance, it exhibited a significant difference with $F_{(3,368)} = 9.73$, $P<0.05$ in terms of household land size and use. The farm size owned ranged from a Mean±std. dev of 5.07±5.06 to 12.45±16.86 translating to an overall Mean±std. dev of 7.24±8.90 acres per household, which was slightly lower than the county average of 5.14 acres (County Government of Elgeyo Marakwet, 2018). However, the Elgeyo farm sizes were higher compared with a study in East Mau that estimated the land owned per household ranging between 1.7 and 2.5 acres (Langat et al., 2016). Despite the varied land sizes, the households use respective parcels for various land uses, whereby crop production took the largest chunk portion of the household land with a Mean±Std. Dev of 36.3±24.0 percent of the total land as reported by 94% of the population. Fodder production followed this with a Mean±Std. Dev of 24.8±15.5 percent of the total land as reported by approximately 45% of the population. Other land uses with a significant proportion include fallow land, woodlot, and settlement/infrastructure development. Respectively with a Mean±Std. Dev of 28.4±23.5, 12.7±9.0, and 11.4±11.6 percent of the total land as reported by approximately 2%, 57%, and 49% of the population. The respondent also reported leasing other parcels outside the primary household, including state forest land issued under plantation establishment and livelihood improvement schemes (PELIS), to supplement household production.

The study estimated the average leased land at between a Mean of 1.7±1.74 and 1.3±0.9 acres as reported by approximately 11% and 49% of the population, respectively (Table 4.4). Overall, the study recorded cropland as the most critical land use, with the natural forest as the least land use. This was consistent with county reports (County Government of Elgeyo Marakwet, 2018). Equally consistent with most studies in forest communities, with over 80% of the households reporting cropland as the primary land use (Fekadu et al., 2021). Worth noting, the PELIS system contributes significantly to food production for households around the Elgeyo ecosystems. This is consistent with studies that have cited PELIS as a livelihood program that is synonymous with commercial exotic forest-dominated ecosystems in Kenya (Humphrey et al., 2016; Kagombe, 2014; Waruingi et al., 2021).

Table 4.4: The Elgeyo household land sizes and use across administrative units

Sub-County	Ainabkoi		Keiyo North		Keiyo South		Moiben		Total	
	Mean	**Std. Dev**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**
Land Size (acres)	12.45	16.86	5.07	5.06	6.94	6.82	8.85	4.48	7.24	8.90
N/Forest (%)	11.9	5.5	6.7	5.3	5.2	2.2	7.1	1.7	7.1	4.7
Woodlot (%)	14.8	6.4	9.8	6.0	13.6	11.0	17.6	8.8	12.7	9.0
Food Crop (%)	41.6	19.7	34.5	24.0	35.4	25.8	40.4	15.3	36.3	24.0
Cash crop (%)	14.5	10.0	5.2	2.7	8.5	5.8	-	-	9.8	7.3
Settlement (%)	14.8	9.9	4.5	5.1	12.2	12.1	-	-	11.4	11.6
Pasture (%)	28.7	15.2	15.8	15.9	25.2	15.1	26.4	15.4	24.8	15.5
Fallow land (%)	-	-	25.0	-	28.9	25.7	-	-	28.4	23.5
Lease (acres)	2.90	4.01	0.79	0.33	1.71	1.18	-	-	1.72	1.74
PELIS (acres)	1.17	0.68	1.51	0.96	0.83	0.71	1.07	0.45	1.25	0.91

The assessment of the land use around the Nyambene ecosystem recorded nonsignificant differences across administrative units. The study estimated the overall mean at 1.6±10.1 acres per household as reported by 99% of the population. The findings show that household plot ownership is relatively even throughout sub-counties but smaller than land sizes around the Elgeyo ecosystem. This land holding size is much smaller than what the county reports, with holdings ranging from 4.45 acres to 44.5 acres (Meru County Government, 2018).

The study attributes this to a couple of factors, including an unwillingness to disclose the actual household landholding and pronounced land fragmentation associated with high population density within forested ecosystems. However, the forest community reported similar land uses whereby crop production recorded the highest proportion of land use at a Mean of 73.5±21.0 percent of the total household land. Infrastructural development and woodlot land followed this, with a mean of 12.7±14.1 and 10.5±13.2 percent of the total land respectively as reported by 99% of the population. Besides the principal household parcel, approximately 11% of the population reported leasing land elsewhere with an overall mean of 0.7±0.52 acres aimed at supplementing household production (Table 4.5). The findings suggest that the primary household land use is crop production consistent with forest communities and rural households, which largely utilise their household land for crop production (Beckline et al., 2022; Fekadu et al., 2021; Shahi et al., 2022; Yego et al., 2021).

Table 4.5: The Nyambene household land sizes and uses across administrative units

Sub-County	Igembe Central		Igembe South		Tigania Central		Tigania East		Total	
	Mean	**Std. Dev**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**
Land Size (Acres)	0.9	0.8	1.1	0.7	2.5	18.3	1.4	1.6	1.6	10.1
Woodlot (%)	12.3	18.5	6.4	9.0	7.6	6.4	14.0	15.5	10.5	13.2
Crops (%) N=400	69.5	26.0	78.3	24.6	72.1	18.2	73.7	19.0	73.5	21.0
Settlement (%)	12.9	16.3	11.2	19.3	15.0	13.6	11.4	10.3	12.7	14.1
Fallow (%)	5.7	9.5	2.6	4.3	2.8	4.0	0.7	3.2	2.4	5.2
Lease (acres)	0.6	0.29	1.5	0.71	0.5	0.38	0.8	0.56	0.7	0.52

4.1.4 Livestock Husbandry

The community living around the Elgeyo ecosystem also practice livestock production, as reported by the majority (87%) of the total households. The mean flock size of livestock per household varies significantly across the administrative unit. Except for goats and sheep, other livestock recorded a significant difference with the Kruskal-Wallis test exhibiting $P<0.05$. The study estimates the average head of livestock per household at about 5.3±5.2, 4.82±3.48, 1.2±0.46, 5.2±3.7, 16.8±18.8 for cattle, goats, donkey, sheep, and poultry, respectively. The study estimates the TLU at a mean of 8.1±3.0, which varied significantly across the administrative units with Kruskal-Wallis test $X^2_{(3)} = 12.04$, $p<0.05$ (Table 4.6). The average TLU was higher compared to a study in East Mau, which estimated the mean TLU at 4.9 per household (Langat, 2016). We could attribute this to the land size and tenure system and mode of residency. For instance, a majority of Mau inhabitants are immigrants where the government excised part of the East Mau forest in the 1990s and settled communities in early 2000. It allocated each household a maximum of 2.5 acres (UNEP, 2009; UNEP et al., 2006). This is contrary to Elgeyo, who are permanent inhabitants with bigger land sizes, primarily owning the respective parcels. The higher TLU demonstrates a significant contribution of livestock to the livelihoods of the Elgeyo forest community. This is consistent with studies on livestock production and its contribution to household income in rural sub-Saharan Africa (Engida et al., 2015).

Table 4.6: The livestock sizes across administrative units (Elgeyo)

Livestock	Descriptive Statistics					Kruskal-Wallis Test
	%	Mean	Std. Dev	Min	Max	
Cattles	84.6	5.31	5.16	1.0	50.0	$X^2_{(3)}$ =12.893, P=0.005
Goats	23.1	4.82	3.48	1.0	20.0	$X^2_{(3)}$ =12.893, P=0.000
Sheep	46.8	5.22	3.65	1.0	21.0	$X^2_{(3)}$ =7.250, P=0.064
Donkeys	6.5	1.15	0.46	1.0	3.0	$X^2_{(1)}$ =1.889, P=0.163
Poultry	69.7	16.81	18.81	1.0	200.0	$X^2_{(3)}$ =19.061, P=0.000
TLU	86.8	8.1	3.04	0.10	16.0	$X^2_{(3)}$ =12.041, P=0.007

Equally, the Community living around and within the Nyambene ecosystem practices livestock production, though lower average compared to those living around Elgeyo. Different households own different categories of livestock with different scales and sizes, as reported by approximately 84% of the respondents. Animals owned include goats, cattle, sheep, pigs, and poultry. The mean varied significantly for some categories of livestock such as goats, cattle, camels, and poultry, while other categories recorded no significant mean difference across the administrative units (sub-county) around the ecosystem. Estimated the mean livestock per household at 3.0±1.9 (goats), 2.2±1.7 (cattle), 2.5±1.7 (sheep), 1.0±0.0 (donkey), zero (camel), 9.3±1.0 (poultry), 2.5±3.2 (pigs), 7.6±5.03 (rabbit) and 4.5±1.50 (TLU). Although the number of goats, cattle, and poultry varied significantly across the sub-county, the aggregated TLU didn't vary with the Kruskal-Wallis test exhibiting $X^2_{(3)} = 7.458$, $P>0.05$ (Table 4.7). This was lower than the TLU recorded in Elgeyo TLU but consistent with a study in East Mau (Langat, 2016). It primarily attributed this to smaller land size and the fact the Ameru is a small-scale pastoral community, unlike the Kalenjin that dominates the East Mau and Elgeyo.

Table 4.7: The livestock sizes across administrative units (Nyambene)

Livestock Category	Descriptive Statistic					Kruskal-Wallis Test
	HH %	Mean	Std. Dev.	Min	Max	
Goats	45.0	2.86	1.93	1.00	16.00	X^2 (3) =27.89, P=0.000
Cattle	62.7	2.18	1.66	1.00	10.00	X^2 (3) =23.575, P=0.000
Sheep	13.7	2.53	1.73	1.00	9.00	X^2 (3), P=0.393
Donkey	1.2	1.00	-	1.00	1.00	X^2 (1) =0, P=1
Poultry	68.2	9.32	11.56	1.00	100.00	X^2 (3) =22.051, P=0.000
Pigs	1.5	2.50	3.21	1.00	9.00	X^2 (2) =3.333, P=0.189
Rabbit	1.2	7.60	5.03	2.00	15.00	X^2 (2) =3.00, P=0.223
Pets	5.0	1.50	0.76	1.00	3.00	X^2 (2) =3.260, P=0.196
TLU	84.8	4.5	1.50	0.01	9.20	X^2 (3) =7.458, P=0.059

4.1.5 Household income sources and levels

The community living within and around the Elgeyo ecosystem depends on several livelihoods and income options, including but not limited to farming, livestock production, and forest product income. However, income varied significantly across administrative units $F_{(3,373)}$ =104.195, P<0.05, where households within Keiyo North recorded the highest annual income with a mean of KES 315,517.54±289,296.34 compared to the Ainabukoi Sub-county that recorded a mean of KES 225,454.91±318,495.16. The study estimates were slightly higher in comparison with a study of East Mau that estimated the household income between KES 170,000/ and

260,000/ (Langat et al., 2016). This translates to between KES 26,293.10 (USD 245.73) and 18,787.83 (USD 175.58) monthly income for the highest and lowest, respectively. These estimates fall into the lower-income earner category, according to Kenya's economic survey of 2017 (Cytonn, 2023). The primary income contributed differently to household income, with agricultural production as the principal contributor at aggregate estimates of approximately 58% of the total income. Business enterprise and forest income followed this at 14%, and 5% respectively of the total income (Table 4.8). The agricultural contribution to household income is consistent with other studies that have reported farming and livestock as the primary contributor to forest communities' income (Kamanga et al., 2009; Mamo et al., 2007). Notably, the contribution of most household programs to the aggregated income varies significantly, while some were uniform across sub-counties around the ecosystems.

Table 4.8: The primary household income sources (Elgeyo)

Income Source	%	Total Mean	Std. Dev	Kruskal-Wallis Test	(%)
Farming	91.4	89,736.07	187,388.95	$F_{(3,341)}$ =37.19, P<0.01	32.37
Livestock Sale	88.7	18,423.44	30,323.48	$F_{(3,331)}$ =85.93, P<0.01	6.65
L/Products Sale	87.4	54,201.68	65,396.55	$F_{(3,326)}$ =63.82, P<0.01	19.55
Remittance	90.1	6,198.99	19,718.97	$F_{(3,336)}$ =33.267, P<0.01	2.24
Forest Product	90.1	12,501.98	68,878.16	$F_{(3,336)}$ =0.719, P=0.869	4.51
Pension	90.1	357.14	4,873.67	$F_{(3,336)}$ =0.541, P=0.910	0.13
Rental	90.1	599.70	7,352.82	$F_{(3,336)}$ =2.769, P=0.429	0.22
Business	88.7	38,821.45	81,390.00	$F_{(3,331)}$ =177.80, P<0.01.	14.01
Expenditure	100	147,145.2	259,882.7	$F_{(3,373)}$ =124.43, P<0.01.	53.09
Total Income	91.2	277,179.8	1,234,535.4	$F_{(3,373)}$ =104.20, P<0.01.	100.00

The contribution of household livelihood programs quoted to aggregated income in Nyambene varies significantly across administrative units. This was as exhibited with the Kruskal-Wallis test exhibiting a significant mean difference with $F_{(3,363)}$ =73.646, P<0.05. The combined income across administrative units records Igembe South with the highest mean of KES 207,070.42±211,006.59, while Tigania Central sub-county recorded the lowest with a mean of KES 69,391.25±140,931.24. The estimates translate to a monthly household income of KES 17,255.87 (USD 161.27) and 5,782.58 (USD 54.04) for the highest and lowest averages, respectively (Table 4.9). This is lower in comparison with the Elgeyo estimates and falls within the low-income bracket reported at over KES 24,000 by Cytonn (2023). This would be attributed the difference in a couple of parameters, including landholding size, and failure to fully disclose the actual income and sources.

Similarly, the study also recorded varied livelihood options a which contribute differently to the overall household income. For instance, farming was the highest contributor with approximately 49% followed by livestock production and small business enterprise at approximately 19% and 17% respectively. Cumulatively, the other programs contributed less than five percent of household income. The study findings were consistent with similar studies that reported agricultural production as the principal contributor to forest community household income (Kalaba et al., 2013; Mamo et al., 2007). That notwithstanding, the contribution of the various programs quoted confirms the diversification of livelihood options consistent with what the literature has advocated (Kamanga et al., 2009; Sunderlin et al., 2005).

Table 4.9: The primary household income sources (Nyambene)

Nyambene Income/Sub-County	**Valid Percent (N=402)**	**Total**		**Kruskal-Wallis**	**%**
		Mean	**Std. Dev**		
Farming	94.8	68,199.21	90,397.05	$F_{(3,381)}$ =81.645, P=0.00	48.87
Livestock Sale	94.8	10,846.09	32,290.13	$F_{(3,381)}$ =42.387, P=0.000	7.77
L/products	97.0	16,512.67	51,258.63	$F_{(3,390)}$ =90.547, P=0.000	11.83
Remittance	98.5	2,396.46	26,267.79	$F_{(3,396)}$ =17.887, P=0.000	1.72
Pension	94.8	2,005.25	23,544.13	$F_{(3,381)}$ =3.074, P=0.380	1.44
Lease land	94.0	158.73	3,086.07	$F_{(3,378)}$ =4.478, P=0.214	0.11
Rental	88.1	655.37	5,243.09	$F_{(3,354)}$ =2.169, P=0.538	0.47
Business	99.0	23,531.41	99,650.82	$F_{(3,398)}$ =31.398, P=0.000	16.86
Forest Income	99.5	977.50	7,971.25	$F_{(3,400)}$ =9.420, P=0.024	0.70
Expenditure	91.5	125,691.03	183,935.81	$F_{(3,368)}$ =41.309, P=0.000	90.07
Gross Income	90.3	139,552.89	249,228.01	$F_{(3,363)}$ =73.646, P=0.000	100.0

Overall, the accumulated household income shows a majority of the population around Elgeyo earns over KES 75,000, which translates to over two dollars a day for a majority of the population. Although a majority recorded modest expenditure income, it was almost double for households with lower income (Figure 4.1). The scenario could either result from some households not being willing to disclose their household earnings or getting support from other sources to support the respective household expenditure. However, a majority (≈53%) of the Nyambene households earn less than KES 75,000/ and approximately 25% of the population reported in the second-level income band (Figure 4.2). This suggests that most of the population earns below the World Bank poverty line of USD 2.15 per day (The World Bank, 2022). Notably, a majority of the household have more expenditure than the household income consistent with most rural forest communities (KWTA, 2020a). The finding suggests that lower-income earners depend on external sources to meet their household expenses. Some of the external sources, though not reported in the study, could include the National Government constituency development fund (NG-CDF) and County Government bursary to fund education and older adults' national government stipend, among other funds.

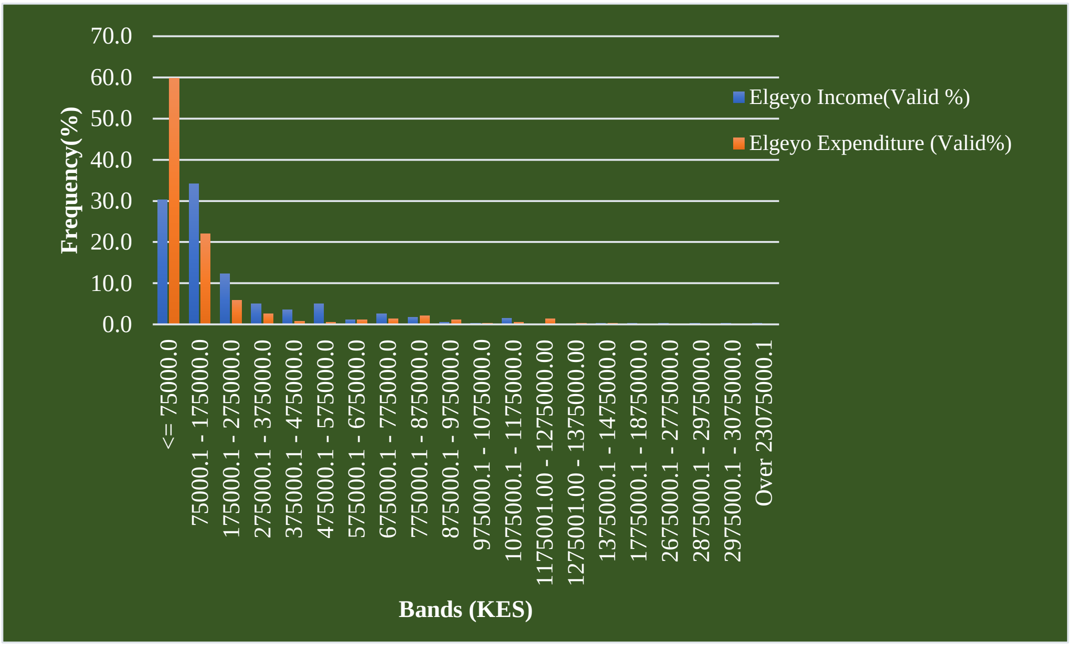

Figure 4.1: Elgeyo Household Income and Expenditure bands

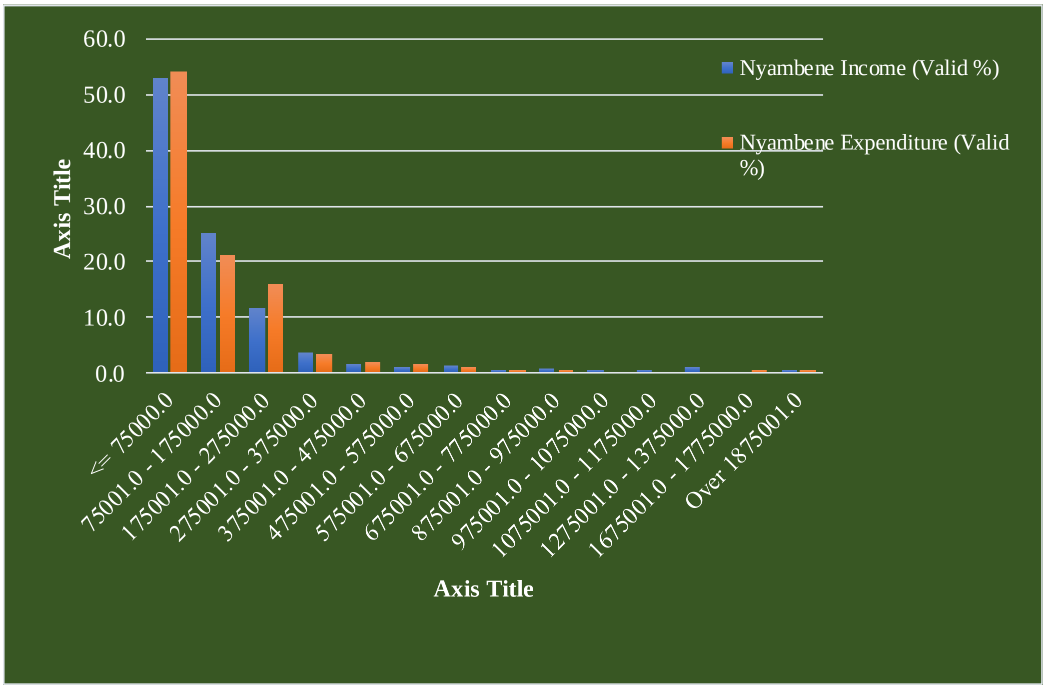

Figure 4.2: Nyambene household income and expenditure bands

4.1.6 Community Perception of the Beneficiaries of ES

The ecosystem's service beneficiaries were divided into five groups by the study, which included the local community, commercial harvesters, forest products traders, government, and foreigners (non-locals). A Likert scale was used to assess the beneficiaries' perceived benefit, with 0 denoting no benefit and 5 representing the highest benefit. The local community's perception of the beneficiaries received different scores from stakeholders. A multi-variance analysis showed a significant difference with Wilks Lambda F (5,12) =2339.957. Friedman's ranked mean placed the community as the highest beneficiaries at 3.4 aggregated mean, while ranked foreigners as the least beneficiaries with ranked Friedman's aggregate mean of 2.7.

Similarly, the Nyambene community perception of the most beneficial stakeholders recorded varied scores, where, for instance, exhibiting a significant difference with $X^2_{(4)}$ =482.712, $P<0.01$. The local community scored as the highest beneficiary with the aggregated Friedman's ranked mean of 3.9 followed by the Government with aggregated ranked mean of 3.7 while scoring foreigners as the least beneficiaries at an aggregated ranked mean of 2.4. Thirty-two percent of beneficiary cohorts can explain the proportion of variance of the dependent variables as shown by Kendall's value of W^a=0.315 (Table 4.10). Overall, the study reveals that the local community scored higher than any other beneficiary in both ecosystems. This is followed by Government while foreigners are rated as the least beneficiary cohort. This concurs with a study in Kenya that reported the local community is the highest beneficiary of forest ecosystem service than in any other category of beneficiaries (MoE & F, 2019a). Equally, KWTA (2020a) reported similar findings, where it scored the community higher than any group of beneficiaries listed.

A multinomial regression test on the Elgeyo perception score as a function of socioeconomic traits (education, age, locality) exhibited significance with $X^2_{(408)}$ =508.482, P<0.01. The model goodness fit exhibited non-significance with a deviance test of $X^2_{(1020)}$ =513.735, P=1.00, suggesting the model fits the data. Equally, it explains approximately 75% of the variance of the perceived benefit scores is associated with the level of education, locality, and age, with pseudo-R-Squared equals 0.749. Likewise, an ordinal regression on the Nyambene perception scores exhibited significance with $X^2_{(15)}$ =311.735, P<0.01. Besides a non-significance on the goodness-of-fit test with Deviance and non-significance test of parallel line with $X^2_{(6095)}$ =1258.788, P=1.000, and $X^2_{(375)}$ =64.575, P=1.000 respectively.

The model can explain approximately 66% of the influence on the perception score attributed to the significant sociocultural and economic factors, as shown by pseudo-R-Squared, equivalent to 0.656. Overall, the findings suggest that the increase in the level of education positively influenced score variance on quoted beneficiaries. Whereas the age of the respondent negatively influenced the score variance, besides the respondent's locality. The influence of education on perception is consistent with a study on forest dependency and participation in common pool resources (Jumbe & Angelsen, 2007), and studies on perception attitudes on benefits towards forest conservation (Kavoi et al., 2019; Ouko et al., 2018). These studies suggested that education, and/or together with socioeconomic factors, such as income, enhance perception scores towards forest conservation attributed to increased conservation awareness. However, the aspect of age was inconsistent with the study by Kavoi (2019) which shows that age does not influence the perception score on the benefit of forest conservation. These inconsistencies would largely be associated with the framing of the questions and method and attributes incorporated in the analysis.

Table 4.10: The summary score of community perception of beneficiaries

ES	%	Com	CH	Trad	Gov	F	Friedman's Test (x^2) df=4
Water	97.6	3.94	3.36	3.28	3.41	3.16	X^2 =190.8, P<0.01, W^a=0.131.
Fuelwood	97.9	3.79	2.90	2.82	3.03	2.47	X^2=238.882, P<0.01, W^a=0.164
Charcoal	96.2	3.51	3.01	3.08	2.94	2.46	X^2=143.389, P<0.01, W^a=0.1
Fodder	95.2	3.91	2.86	2.71	2.99	2.53	X^2=272.845, P<0.01, W^a=0.192
Timber	95.2	3.17	3.06	2.99	3.50	2.28	X^2=190.578, P<0.01, W^a=0.134
Game meat	95.7	3.17	2.78	2.87	3.29	2.90	X^2=51.881, P<0.01, W^a=0.036
Honey	96.0	3.50	2.96	3.05	2.96	2.53	X^2=134.521, P<0.01, W^a=0.094
Farm tools	94.9	3.32	2.95	2.89	2.99	2.84	X^2=45.359, P<0.01, W^a=0.032
Mushrooms	93.6	3.55	3.03	2.95	2.72	2.76	X^2=135.063, P<0.01, W^a=0.095
Twining	93.0	3.19	2.97	2.92	3.07	2.85	X^2=22.853, P<0.01, W^a=0.016
Wild fruits	93.6	3.44	2.93	2.92	2.92	2.80	X^2=76.683, P<0.01, W^a=0.055
Nat/medicine	94.1	3.55	2.92	3.03	2.94	2.56	X^2=145.703, P<0.01, W^a=0.104
Air quality	94.9	3.19	2.91	2.92	3.12	2.86	X^2=33.554, P<0.01, W^a=0.024
Biodiversity	92.0	3.07	2.86	2.98	3.13	2.96	X^2=17.8, P<0.01, W^a=0.013
Cultural	94.1	3.54	2.90	2.89	2.92	2.76	X^2=118.104, P<0.01, W^a=0.084
Disaster	93.3	3.18	2.98	2.93	3.12	2.79	X^2=33.647, P<0.01, W^a=0.024
Tourism	93.6	3.16	2.76	2.83	3.30	2.94	X^2=62.142, P<0.01, W^a=0.045
Elgeyo	**97.2**	**3.42**	**2.95**	**2.94**	**3.08**	**2.73**	$F_{(5,12)}$ =2339.957, P=0.000.
Nyambene	**95.3**	**3.88**	**2.58**	**2.38**	**3.76**	**2.40**	X^2=482.712, P<0.01, W^a=0.315

Com = community; CH = Commercial harvesters; Trad = Traders; Gov = Government; F = Foreigners.

4.1.7 Perception of the Local Community Ecosystem Threats

The study quoted fifteen potential threats to respondents from the two ecosystems, as listed in the table below. On a scale of 0 to 5, with zero being no threat and five being the most threat, the respondent gave varied scores. Friedman's Test exhibited significance with $X^2_{(15)}$ =518.82, P<0.01, Kendall's W^a=0.306 for the Elgeyo and $X^2_{(15)}$

=306.586, P<0.01, Kendall's W^a=0.130 for the Nyambene ecosystems. The score on Elgeyo records high forest product demand, high poverty, illegal harvesting, and fire topped the list in that order. Nyambene scored low staffing, corruption, and pollution as the highest threats to the ecosystem (Table 4.11). This suggests that unique ecosystems face dissimilar forms of threats and pressure. This was consistent with a study on the economics of ecosystem services (MoE & F, 2019a) and on drivers of land use changes (Laurance & Balmford, 2013), both of which reported different degrees of the form of threats unique to the diverse ecosystems. However, high demand for forest products, poverty, and illegal activities appear as the topmost threats to most forested ecosystems (Arima et al., 2011; MoE & F, 2019a; Ndalilo et al., 2020). Worth to note not all respondents acknowledged threats to water catchment ecosystems.

Table 4.11: The summary score of community perception on the ecosystem threats

Elgeyo			**Nyambene**		
Threats	**%**	**Mean Rank**	**Threats**	**%**	**Mean Rank**
High FWP Demand	30.3	10.78	Low Staffing	39.1	11.28
High Poverty	30.3	10.75	Corruption	39.1	10.16
Illegal Harvesting	30.3	10.54	Pollution	39.1	10.14
Fire	30.3	10.41	Poverty	39.1	10.10
Population Pressure	30.3	10.29	Climate Change	39.1	9.58
Corruption	30.3	10.26	Population Pressure	39.1	9.12
Overgrazing	30.3	9.99	Illegal Logging	39.1	8.71
Climate Change	30.3	9.86	State forest dependence	39.1	8.51
State forest dependence	30.3	9.84	Fire Threat	39.1	8.06
Charcoal Production	30.3	8.17	Forest Product Demand	39.1	8.05
Pollution	30.3	6.48	Invasive Species	39.1	7.60
Invasive Species	30.3	6.48	ES Undervaluation	39.1	7.49
Encroachment	30.3	6.01	Encroachment	39.1	7.39
Pest and Diseases	30.3	6.00	Charcoal Production	39.1	6.87
Low Staffing	30.3	5.14	Pest and Disease	39.1	6.63
ES undervaluation	30.3	4.99	Overgrazing	39.1	6.31
X2 (15) =518.82, P=0.000, Kendall's Wa=0.306,			X2 (15) =306.586, P=0.000, Kendall's Wa=0.130.		

The ordinal regression analysis exhibited variability in the respondents' threat score for the Elgeyo, which was influenced by socioeconomic factors. This includes the household tropical livestock unit (TLU), the political jurisdiction (ward level), gender, and the primary occupation among others. Notably, although the individual attribute contribution was non-significant, the aggregation of the factors on the explanatory variables quoted exhibited significant differences. The ordinal regression model confirms the influence of the socio-economic factors quoted as exhibited by $X^2_{(14)}=81.515$, $P<0.01$, showing a significant improvement to the baseline model (intercept). While, the Goodness-of-fit recorded a Pearson $X^2_{(4816)}=3722.518$, $P=1.00$, suggesting that the ultimate model fits the data analysed well. The model can explain approximately 49% of the influence on the perception score attributed to the significant sociocultural, demographic, and economic factors, as shown by Nagelkerke, equivalent to a 0.491. Equally, the test on the parameter line recorded a non-significant value of $X^2_{(574)}=402.677$, $P=1.000$. This suggests a strong relationship between the predictor (socio-economic) factors and the likelihood of the scores recorded falling between the mean range being consistent across the scores.

The developed ordinal regression model exhibited improvement from the baseline model, where the model fitting test shows a significant difference of $X^2_{(16)}=256.374$, $P<0.01$. While the Goodness-fit-test exhibited a non-significance on the Deviance test (Likelihood ratio test) $X^2_{(5060)}=651.577$, $P=1.000$, suggesting the assumed model fits the data well. The analysis also showed approximately 90% influence, with a Pseudo R-Squared of 0.891, suggesting that the socioeconomic and covariate parameters (independent) significantly affect the variance of the perceived mean threat score quoted. The test on the parallel line exhibited non-significance with $X^2_{(736)}=589.194$, $P=1.000$, suggesting that there was consistency and a high likelihood of the model scores using the quoted predictors falling within the mean score range quoted, thus assumptions of proportional odds satisfied. Notably, the age record a positive influence while the household land size recorded a negative influence on the perceived mean threat score quoted. This is consistent with studies on socioeconomic influence on community perception of forestry (Dehghani et al., 2023; Kavoi et al., 2019; Meijaard et al., 2013; Ouko et al., 2018). Where factors such as education enhance awareness, low-income households, and older folks understand more the benefits of the forested ecosystem, thus recording higher scores. This is contrary to what is commonly reported in the literature whereby wealthy households, less educated, and younger folks with lower scores in forestry and conservation (Gouwakinnou et al., 2019; Jha & Gupta, 2021).

4.1.8 Forest Dependency

The study categorised forest dependency into levels that is low and high and used a score of 0.5 as a cut-off point. It categories a score less than 0.5 as a low dependency, while scores greater than or equal to 0.5 as a high dependency as commonly employed in similar studies (Garekae et al., 2017; Jain & Sajjad, 2015). The omnibus tests exhibited significance for Elgeyo with $X^2_{(22)}$ =301.964, P<0.01 while Hosmer and Lemeshow test recorded non-significance with $X^2_{(8)}$ =1.721, P=0.988, both suggesting that the model fit the data analysed. The study can explain the difference in forest dependency by approximately 88% linked to sociocultural attributes, as shown by the Nagelkerke R-Squared score of 0.878 with an accuracy of approximately 94%. The HH expenditure, income, and tropical livestock unit influence the community forest dependency score significantly at 95% CL. For instance, the increase in household expenditure and income level decreases the community forest dependency by a factor of 0.1 and 0.03, respectively. However, an increase in household livestock number increases the dependency by a factor of 3.8. In addition, the establishment of on-farm woodlots decreases the dependency on forest resources by a factor of 3.4 other factors held constant. Other important parameters, though not significant at a 95% confidence level, include the household size and length of residency (Table 4.12). This, therefore, suggests that household income, expenditure, livestock owned, length of residency, and household size influence the forest dependency level, though at different scales and confidence levels. The findings on household size and income bands were consistent with a study by Adam and El Tayeb (2014) and contrary to Garekae et al. (2017) which placed a decrease in dependency, particularly to increase household size. However, factors such as education, age, and distance didn't influence dependency, contrary to what most literature cited to influence forest dependency. For instance, an increase in age and level of education reduces dependency on forest resources (Garekae et al., 2017), although age and distance were inversely proportional to dependency (Najabat et al., 2020). The study could attribute the difference to a couple of factors, including the mode of analysis, method, variables incorporated in the analysis, and the mid-dependency score, among others.

Table 4.12: The summary statistics on forest dependency (Elgeyo)

Elgeyo	B	S.E.	Wald	df	Sig.	Exp (B)	95% C.I. EXP (B)	
							Lower	Upper
Expenditure	-2.346	.837	7.861	1	.005	.096	.019	.494
Income	-3.586	.752	22.722	1	.000	.028	.006	.121
SubCounty			5.189	3	.158			
Ward			5.027	9	.832			
Gender	-.055	.721	.006	1	.939	.946	.230	3.887
Age	.002	.040	.002	1	.963	1.002	.927	1.083
Education	-.224	.361	.384	1	.535	.799	.394	1.623
Length of Residency	-.050	.032	2.408	1	.121	.951	.892	1.013
HH Size	.277	.178	2.406	1	.121	1.319	.930	1.871
Land Size	.058	.080	.538	1	.463	1.060	.907	1.239
TLU	1.329	.284	21.825	1	.000	3.777	2.163	6.596
Distance	-.029	.114	.067	1	.796	.971	.777	1.214
Constant	6.577	2.633	6.239	1	.012	718.440		

Omnibus tests $X^2_{(22)}$ =301.964, P=0.000, Hosmer and Lemeshow test $X^2_{(8)}$ =1.721, P=0.988, Nagelkerke R-Squared equivalent to 0.878, Classification accuracy 94.1%

Similarly, the logistical regression for the Nyambene forest dependency exhibited significant differences in forest dependency as a function of sociocultural and economic attributes. The study confirms this with omnibus tests $X^2_{(22)}$ =159.919, P<0.01, and Hosmer and Lemeshow test $X^2_{(8)}$ =6.368, P>0.05, suggesting that the model fits the data. The study explains the variance in the dependency associated with socioeconomic traits at 74% as shown by Nagelkerke, equivalent to 0.744 with an accuracy of approximately 88%. The analysis established that an increase in household income and distance from the state forest decreases community dependency by a factor of 0.05 and 5.8, respectively. However, an increase in the household size and livestock number increased dependency, consistent with a study in Sudan on forest dependency, a case of the Sarf-Saaid reserve forest (Adam & El Tayeb, 2014). Other important factors, such as age, gender, and education, recorded non-significance at a 95% confidence level (Table 4.13). This was contrary to most literature, for instance, according to Najabat et al. (2020), the increase in the level of education, age, and distance reduces dependency, albeit inversely according to other forest dependency studies (Adam & El Tayeb, 2014; Garekae et al., 2017). Overall, the findings suggest that incomes and distance are critical factors in assessing and modelling forest dependency. The findings were largely in agreement with the growing literature on forest dependency and the respective predictive factors (Htun et al., 2017; Mujawamariya & Karimov, 2014; Ofoegbu et al., 2017; Sapkota & Oden, 2008; Suleiman et al., 2017).

Table 4.13: The summary statistics on forest dependency (Nyambene)

Nyambene	**B**	**S.E.**	**Wald**	**df**	**Sig.**	**Exp (B)**	**95% C.I. EXP (B)**	
							Lower	**Upper**
Expenditure	.093	.114	.669	1	.413	1.098	.878	1.372
Income	-2.911	.532	29.943	1	.000	.054	.019	.154
SubCounty			4.060	3	.255			
Ward			10.325	7	.171			
Gender	.636	.615	1.068	1	.301	1.889	.566	6.309
Age	-.027	.024	1.206	1	.272	.974	.928	1.021
Education	-.389	.399	.947	1	.330	.678	.310	1.483
Birth	.826	.920	.806	1	.369	2.284	.376	13.856
HH Size	.315	.138	5.232	1	.022	1.371	1.046	1.796
Land Size	-.294	.558	.277	1	.599	.746	.250	2.226
Woodlot	.026	.034	.584	1	.445	1.026	.960	1.097
Cropland Size	-.031	.020	2.313	1	.128	.969	.931	1.009
TLU	1.709	.339	25.463	1	.000	5.523	2.844	10.726
Distance	-1.320	.343	14.848	1	.000	.267	.136	.523
Constant	21.377	10986.184	.000	1	.998	1.9x10^9		

Omnibus tests $X^2_{(26)}$ =159.919, P=0.000, Hosmer and Lemeshow test $X^2_{(8)}$ =6.368, P=0.606, Nagelkerke R-Squared equivalent to 0.744, Classification accuracy 86.8%

4.2 The Economic Valuation of Ecosystem Goods and Services

4.2.1 Direct Use Values

4.2.1.1 Harvest and Sourcing of Direct-Use Ecosystem Services

The study found that households within and around the Elgeyo and Nyambene harvest/collect direct use/ forest provisioning ecosystem services (FPES) at different frequencies and scales. This is as reported by 72% and 14% for Elgeyo and Nyambene ecosystems, respectively. From the seventeen ecosystem services quoted, fuelwood recorded the highest harvest frequency in both the ecosystems at 97% and 79% for Elgeyo and Nyambene, respectively. The least harvested forest products in both ecosystems included thatch grass and game meat for Elgeyo and thatch grass, marram, reeds, mushroom, and game meat for the Nyambene. Notably, the harvest of forest products was higher in Elgeyo than in Nyambene for most of the products, as reported by over 70% of the households (Table 4.14). The highest harvest frequency of fuelwood recorded in both ecosystems is consistent with studies of forest provisioning services and valuation. For instance, according to a study by Dhyani and Dhyani (2016) in the upper Kedarnath Valley, Garhwal, India, forest community source 95% of their domestic energy from fuelwood. This was equally consistent with a study by Langat (2016) in East Mau that established over 90% of the forest community collects fuelwood as their primary source of cooking energy and heating. The study could attribute this to, among others, a higher poverty index and a lack of affordable alternative energy sources, in concurrence with a study on biomass use in Kenya by Mugo and Gathui (2010). Overall, although the harvesting frequency differs in the two ecosystems, the scale of frequency on most products in Elgeyo reveals the significance of forest provisioning services on household income and livelihoods. This is consistent with similar studies showing to contribute over 30% to household income and livelihoods (Kalaba et al., 2013; Najabat et al., 2020).

Table 4.14: Household Harvest of Direct Use Ecosystem Services

Ecosystem Services	Elgeyo			Nyambene		
	Harvest/Collection Frequency			Harvest/Collection Frequency		
	% (N=373)	No	Yes	% (N=402)	No	Yes
Fuelwood	99.46	0.03	0.97	96.02	0.21	0.79
Timber	99.20	0.05	0.95	95.02	0.93	0.07
Charcoal Production	99.46	0.34	0.66	95.52	0.88	0.12
Honey	99.20	0.16	0.84	96.02	0.87	0.13
Natural Medicine	99.20	0.02	0.98	95.27	0.96	0.04
Fencing Poles	99.20	0.05	0.95	95.27	0.90	0.10
Building Poles	99.46	0.07	0.93	95.27	0.96	0.04
Thatch Grass	100.00	1.00	-	94.78	1.00	-
Wild Fruits	98.93	0.02	0.98	95.02	0.61	0.39
Fodder	99.46	0.27	0.73	95.27	0.77	0.23
Farming Hand Tools	98.39	0.06	0.93	96.02	0.76	0.24
Marram	99.20	0.21	0.79	95.52	1.00	-
Quarry Stones	99.46	0.14	0.86	95.77	0.97	0.03
Mushroom	99.20	0.44	0.56	95.27	0.99	0.01
Reeds	98.66	0.55	0.45	96.02	0.99	0.01
Game Meat	100.00	1.00	-	95.02	0.99	0.01
Aggregate	99.28	0.28	0.72	95.44	0.86	0.14

The forest community harvested and collected diverse forest products from different sources, including private farms, neighbourhood farms, the local market, and/or the state-managed forest. The aggregate values show a majority (54%) of Elgeyo sources these products from local traders, while 29% are from private farm sources. However, the Nyambene community primarily sources the products from private farms as reported by approximately 46% followed by public forest sources as reported by 31% of those harvesting. Narrowing to the state-managed forest, fuelwood is more highly reported than any other product in the state-managed forest as reported by approximately 38% and 56%, respectively. The other significantly reported product sourced from Elgeyo state-managed forest is natural medicine and mushroom, as reported by 12% and 17% respectively. While in Nyambene, the second most common product sourced is natural medicine and honey, as reported by 67% and 40% of those harvesting the products (Table 4.15).

Overall, the study established that a majority of the respondent source forest products outside state-managed forests, including local markets and private farms. This is contrary to a study in East Mau that suggested a majority of forest community source products from state-managed forests (Langat, 2016). Although it sounds positive in reducing pressure on state-managed forests, this could be the contrary because of the factor of fear of victimisation. Forest resource assessment surveys often face a common problem where respondents are afraid to disclose their harvesting locations, especially when conducted by government officials (Eregae et al., 2023; MoE & F, 2019a).

Table 4.15: Score of Household Sourcing of Direct Use Ecosystem Services

ES	Elgeyo Ecosystem					Nyambene Ecosystem				
	%	**OF**	**NF**	**LM**	**PF**	**%**	**OF**	**NF**	**LM**	**PF**
Fuelwood	97	0.54	0.04	0.04	0.38	76	0.38	0.01	0.06	0.56
Timber	95	0.62	0.10	0.24	0.03	6	0.77	0.15	0.04	0.04
Charcoal	66	0.36	0.07	0.56	0.02	11	0.15	0.02	0.83	-
Honey	84	0.12	0.03	0.78	0.07	12	0.27	-	0.35	0.39
N/Med	98	0.04	0.04	0.80	0.12	4	-	-	0.33	0.67
F/Poles	95	0.62	0.10	0.21	0.07	9	0.45	0.42	0.03	0.11
B/Poles	93	0.32	0.08	0.56	0.04	4	0.65	0.29	-	0.06
W/Fruits	98	0.17	0.02	0.78	0.02	37	0.63	0.01	0.12	0.24
Fodder	73	0.62	0.02	0.29	0.07	22	0.74	0.18	0.01	0.07
H/Tools	93	0.08	0.01	0.88	0.03	23	0.50	0.14	0.28	0.08
Marram	79	0.08	0.30	0.58	0.04	0	-	-	-	-
Q/Stones	86	0.10	0.24	0.62	0.04	2	0.10	-	0.80	0.10
Mushroom	56	0.24	0.03	0.56	0.17	0	-	-	-	0.0
Reeds	45	0.01	0.15	0.82	0.01	0	-	-	-	0.0
G/Meat	0	-	-	-	-	1	-	-	0.50	0.50
Aggregate	**72**	**0.29**	**0.09**	**0.54**	**0.08**	**13**	**0.46**	**0.07**	**0.16**	**0.31**

OF = Own-farm; NF =Neighbour's farm; LM = Local market; PF =Public forest.

4.2.1.2 Annual household harvested quantities and economic values

The study records that apart from water resources reported by all the households, fuelwood is the second most harvested product at 67% and 76% of the Households in Elgeyo and Nyambene, respectively. In Elgeyo, a household harvests 5 back-loads of fuelwood per week, while in Nyambene, a household harvests 3 back-loads per week equivalent to 264 and 165 head loads per household annually, respectively. This translates to 8,712kg and 5,445kg, respectively based on Maua et al. (2019), back-load weight of about 33kg. This was higher for the case of Elgeyo and lower for the case of Nyambene compared with a study in the South Nandi forest reserve that estimated average household fuelwood use at about 7300kg annually.

The study findings were higher compared with a study in East Mau estimated 4,026kg (122 head loads) per HH annually (Langat, 2016), and a study in the Eastern Himalayan region of India which estimate fuelwood use per HH at between 3,581kg and 4,867kg (Saha & Sundriyal, 2012). Likewise, a study on rural farms in Rwanda estimated household firewood harvest at about 5200kg annually (Ndayambaje et al., 2012). Overall, the fuelwood harvest estimates were within range in comparison with a study in Mau complex, Cherangany, and Mt Elgon that ranged between 5,214kg (158 back-loads) and 9,504kg (288 back-loads) (MoE & F, 2019a). The study could attribute the discrepancies to a couple of factors, including the purpose of harvest, mode of transport, distance to the forest, and household needs, among others. Other significant products, though some less than 10% of the household report, include charcoal, fodder, and farm tools. The study estimated four 90kg gunny bags of charcoal to be harvested by 7% of households in Elgeyo. Although households reported other products in the Elgeyo ecosystem, were not as frequent.

However, in Nyambene, the study records a couple of products, including wild fruits, farm tools, fodder, honey, and charcoal that were harvested by at least 5% of the households. And, the quantities harvested by households vary across administrative units (Appendix II). Notably, the community mainly uses forest products sourced from the two ecosystems domestically, as reported by approximately 78% and 94% for Elgeyo and Nyambene, respectively. Suggesting that they set only a small portion of the community harvests for sale. This was consistent with most studies on the valuation of ecosystem services that commonly report that households primarily used forest harvests for domestic and that a modest proportion is for sale (MoE & F, 2019a).

Using kerosene unit price (KES 157) as a proxy, the study estimates the fuelwood value at KES 37,907.44 and 26,059.64 for the Elgeyo and Nyambene ecosystems, respectively. This was slightly lower compared with a study in the Marsabit reserve on the economic valuation of non-timber forest products that estimated wood fuel at KES 55,296 per household annually (Odiakha, 2015). The estimates were slightly higher compared to a study in Kano Plains in Nigeria on an analysis of the economy of non-timber forest products, which estimated fuelwood value at KES 25,000 per household annually (Suleiman, 2017). The study attributes the discrepancies chiefly to diverse unit prices, quantity harvested per household, and, to some extent, the valuation method. Charcoal production was another critical product as reported by 7% of the population (Elgeyo) estimated at bout 216 (90kg gunny bags) with a monetary value of KES 183,788.90 per household per year. This was slightly higher compared to the East Mau study, which estimated the charcoal value at 144,156 per household annually (Langat, 2016). Other significant non-timber forest products harvested and reported by over 10% of the households in Nyambene include fodder and farming tools (Table 4.16).

Table 4.16: Household Annual Harvest and Economic Values

ES	Elgeyo		Nyambene	
	Quantities	**Values (KES)**	**Quantities**	**Values (KES)**
Fuelwood	264.42±56.36	41,473.12±8,848.2	165.98±121.1	26,059.64±19,012
Timber	1,200.00	42,000.00	393.33±943.6	14,946.67±35,857
Charcoal	216.22±703.9	183,788.89±598,328.5	29.67±30.00	25,221.8±25,499.1
Honey	129.00±274.5	64,500.00±137,232.08	67.23±170.73	33,615.38±85,363
Medicine	52.30	7,845.00	84.00±103.03	12,600.00±15,454
F/Poles	547.07±990.5	82,060.00±148,576.74	151.00±210.7	22,650.00±31,608
B/Poles	200.00±130.9	50,000.00±32,732.68	3.00±1.41	750.00±353.55
W/Fruits	52.30	2,615.00	9.44±24.99	472.22±1,249.25
Fodder	3,003±1,019.	30,035.14±10,190.04	5,617±9,696	56,172.21±96,959
Tools	3.50±1.91	175.00±95.74	1.12±0.38	55.77±18.93
Q/Stones			4,000±2,646	140,000.0±92,601
Marram	133.33±57.74	133,333.33±57,735.03	-	
Mushroom	72.42±58.62	10,862.31±8,793.44	666.00±687	99,900.0±103,096
G/Meat	-		144.00±135.8	21,600.0±20,364.7

4.2.1.3 Water Resources

4.2.1.3.1 Domestic water use

The survey confirms that the two ecosystems are the source of domestic water supply, particularly to the local community as reported by 75% and 35% of the households for the Nyambene and Elgeyo, respectively. However, it recorded varied primary sources where, for instance, the households around the Nyambene majority (59%) reported sourcing water through piped infrastructure, while the Elgeyo (53%) sourced from boreholes. Those involved in collecting water are mainly women and children, as confirmed by over 60% in each of the two ecosystems (Table 4.17). Estimated the household consumption at a mean± std deviation of 89.44±26.66 and 87.14±40.73 litres daily for the Elgeyo and Nyambene, respectively. The estimates are consistent with a study in East Mau by Langat (2016) which estimated daily domestic water consumption at 87.8 litres per household. However, the estimates were lower compared with a study on per capita domestic water consumption, which estimated it at 119 litres per household within prudential estates, though higher middle-class estates at 58 litres (Otieno, 2005). The study associates the difference in the level of resource utility based on family size, social class, distance to the source, and access rights.

Table 4.17: Responses and Frequency on Water Sources and Involvement

Parameters	Variables	Elgeyo Frequency (%) N=373	Nyambene Frequency (%) N=402
Primary Water Source	Borehole	53.12	12.34
	Piped/Tap	36.86	59.38
	Water pans/dams	0.27	0.26
	Stream/river	9.76	24.94
	Spring	-	3.08
Forest Water Source	No	64.84	24.61
	Yes	35.16	75.39
Who is involved?	Men	38.87	15.92
	Women	58.18	67.91
	Children	79.09	62.69
D/Water	Quantity	32,743±9,760	31,929±14,910

4.2.1.3.2 Livestock water use

The study confirms that the ecosystem is critical for livestock water provision primarily based on the respective household livestock size and demand. There's significant variation in the water demand for the two ecosystems across administrative units. The literature reports livestock water demand ranges between 20 and 50 litres per tropical unit. It attributes this to, among others, the species and bread, ambient temperature, livestock activity, age, reproduction stage, and foliage moisture content (IWMI, 2007; Parker & Brown, 2003; Sileshi et al., 2003; Ward & McKague, 2007). In that regard, using the average voluntary tropical livestock water demand estimates of 26.8 litres, the aggregate value shows that the daily household's livestock water demand was about 177 litres and 59 litres in Elgeyo and the Nyambene ecosystems, respectively (Table 4.18). This translates to 64,793 litres and 21,755.04 litres of water for livestock use per household annually, respectively. The estimates were higher for the case of Elgeyo but lower for the case of Nyambene in comparison with a study in East Mau Langat (2016) that estimated annual livestock consumption at about 40,000 litres per household. The study attributes the difference to the number of livestock and the unit water demand applied in the study, where a reference study recorded an average TLU of 4.9 and a unit water demand of 22 litres.

Table 4.18: Daily Household's Livestock Voluntary Water Demand

Livestock	Voluntary water demand (Litre) [(a)]	Elgeyo		Nyambene	
		Mean TLU	HH Water Demand (Litre)	Mean TLU	HH Water Demand (Litre)
Cattle	26.7	6.38	170.08	2.62	69.84
Goats	36.7	0.48	17.66	0.29	10.47
Sheep	36.7	0.52	19.13	0.25	9.27
Donkey	27.0	0.46	12.46	0.40	10.80
Camel	20.7	-	-	-	-
Poultry	19.7	0.17	3.31	0.09	1.83
Pig	27.0	-	-	0.75	20.25
Rabbit	19.7	-	-	0.15	2.99
TLU	26.8	6.62	177.03	2.22	59.44

(a) *Voluntary livestock water demand* (IWMI, 2007; Sileshi et al., 2003)

4.2.1.3.3 Annual water use and valuation

The study estimates daily household water consumption for domestic (89.44 litres and 87.14 litres) and livestock (177.03 litres and 59.44 litres) for the Elgeyo and Nyambene, respectively. This translates to 4.3 million M^3 and 5.5 million M^3 (domestic) and 8.03 million M^3 and 3.6 million M^3 (Livestock), respectively (Table 4.19). Based on a surrogate unit cost of KES 0.23 per litre, the study estimates the aggregated value for domestic and livestock water at KES 1.0 billion (USD 9.4 million) and KES 1.3 billion (USD 12.13 million) (domestic); and KES 1.9 billion (USD 17.6 million) and KES 751 million (USD 7.02 million) (livestock). The Elgeyo households had an annual estimate of KES 7,666.29 (USD 71.65) and domestic households had 7,469.14 (USD 69.81). Livestock households in Nyambene had KES 14,415.30 (USD 134.72) and 4,321.97 (USD 40.39) annually. The findings are higher for the case of Elgeyo and Lower for the case of Nyambene compared with Mau East study findings that recorded per household annual livestock water demand of about 40,352 litres (Langat, 2016). The study primarily attributed the discrepancy to the number of beneficiaries and respective tropical livestock per study area.

Table 4.19: Domestic and Livestock Water Consumption Valuation

Water Abstraction	Elgeyo		Nyambene	
	Domestic	Livestock	Domestic	Livestock
HH Daily demand (litres)	89.44	177.03	87.14	59.44
Number of beneficiaries	130,597	130,597.00	173,743.00	173,743.00
Proportion (%)	1.00	0.95	1.00	0.85
Total Daily demand (m^3)	11,680.60	21,963.61	15,139.97	8,760,634.95
Annual water demand (m^3)	4,275,098	8,038,680.37	5,541,227.2	3,206,392,391.5
Boreholes yield m^3 hr-1	3.50	3.50	3.50	3.50
Maximum hours	10.00	10.00	10.00	10.00
Borehole extraction (m^3)	35.00	35.00	35.00	35,000.00
Annual borehole (m^3)	12,810.00	12,810.00	12,810.00	12,810,000.00
Annual Borehole eq.	333.73	627.53	432.57	250.30
Cost of Sinking BH (KES)	3,000,000	3,000,000	3,000,000	3,000,000
Aggregated Value (KES)	1billion	1.9billion	1.3billion	751million
Annual HH Value (KES)	7,666.29	14,415.30	7,469.14	4,321.97

4.2.1.4 Valuation of Free-Range Livestock Grazing

As aforementioned, the community around the two ecosystems is agropastoral and thus relies on the two ecosystems for livestock grazing. The study found that communities graze livestock on free range in the two ecosystems as reported by approximately 19% and 8% of the population for Elgeyo and Nyambene, respectively. Based on the household livestock size and the respective weight, the study estimates average TLU at 8.01 and 4.55 for Elgeyo and Nyambene, respectively. The study equates this to dry matter demand of 37.25kg and 21.16 kg per household per day, respectively (Table 4.20). Equally, this corresponds to 13,633 kg and 7,744.56kg of fodder demand per household per year, respectively. Using the unit cost of fodder alternative (hay) which is estimated at KES 250 per 25kg bale. In that regard, the study estimates the grazing value at KES 136,330.00 (USD 1,274) and KES 77,445.60 (USD 723.79) per household per year, for Elgeyo and Nyambene, respectively. This was higher compared to the study in the south Nandi forest reserve that valued hay equivalent grazing at between KES 32,836.87 (USD 306.89) and 43,782.49 (USD 409.18) HH^{-1} $year^{-1}$ (Maua et al., 2019). Equally, the study estimates were higher compared to a study in East Mau that estimated the grazing value at between 13,832 (USD 129.27) and 20,800 (USD 194.39) HH^{-1} $year^{-1}$ (Langat et al., 2016). The study attributes the discrepancies chiefly to the size of household TLU and the unit price of either a surrogate employed or the actual product.

Table4.20: Household Tropical Livestock Dry Matter Demand (DMD) Statistics

Descriptive Statistics	Average weight (Kg)	TLU Factor	DMR	Elgeyo			Nyambene		
				livestock	TLU	DMD (Kg)	Livestock	TLU	DMD (Kg)
Cattle	300	1.2	0.025	5.32	6.38	23.92	2.18	2.62	9.82
Goats	25	0.1	0.038	4.82	0.48	2.75	2.86	0.29	1.63
Sheep	25	0.1	0.035	5.22	0.52	2.74	2.53	0.25	1.33
Donkey	100	0.4	0.030	1.15	0.46	2.08	1.00	0.40	1.80
Camel	350	1.4	0.030		-	-	-	-	-
Poultry	2.5	0.01	0.030	16.81	0.17	0.76	9.32	0.09	0.42
Pigs	75	0.3	0.030		-	-	2.50	0.75	3.38
Rabbit	5	0.02	0.030		-	-	7.60	0.15	0.68
TLU	250	1	0.031		8.01	37.25		4.55	21.16

DMD (Dry matter demand); DMR (Dry matter ratio); and TLU (Tropical Livestock Unit)

4.2.1.5 Cultural, Spiritual, and Recreational Services

The study determined cultural and recreational services using the number of beneficiaries, visit frequency, and the mean maximum willingness to pay. The study establishes that visitation frequency for cultural and recreational services to vary across administrative units. For instance, in Elgeyo ecosystems, the Kruskal-Wallis test exhibited a significant difference with $F_{(2,27)} = 19.489$ $P<0.01$. Friedman's rank test placed households from Keiyo South with the highest number of visits and Moiben with the least number of visits. Similarly, the visitation frequency within the Nyambene varied across sub-counties, whereby the Kruskal-Wallis test exhibited significance of $F_{(3,\ 217)}$ =37.602, P<0.01. Households in Tigania East recorded the highest level of visitation, while Igembe Central recorded the least visitation.

The study estimated that 20% of the total households in Elgeyo visit the ecosystem for cultural/spiritual purposes, whereas 54% of the total households in Nyambene visit the ecosystem for the same purposes. The study estimated average length of visitation for cultural/spiritual purposes is 6.08±13.72 days in Elgeyo and 32.78±36.49 days in Nyambene. According to Eregae et al. (2021), the maximum willingness to pay (WTP) for cultural values is USD 7.4±0.3 HH^{-1} $year^{-1}$. In that aspect, the study estimates cultural/spiritual values at KES 128.1 million (USD 1.2 million) and 2.9 billion (USD 22.8m) annually for the Elgeyo and the Nyambene ecosystems, respectively (Table 4.21). This translates to KES 1,184.83 (USD 11.07) and KES 80,800.00 (USD 755.14) $ha^{-1}year^{-1}$ for Elgeyo and the Nyambene ecosystems, respectively. These values were higher than the unit estimates of a study in the Mau Complex and Elgon, at 243.73 (USD 2.28), 400.85 (USD3.75) (Langat et al., 2021). However, the Elgeyo cultural/spiritual values were consistent with a study in Cherangany valued at KES 827.26 (USD7.73) $ha^{-1}year^{-1}$ (MoE & F, 2019a).

Equally, for recreational services, the visitation frequency varies across the sub-counties, whereby the Kruskal-Wallis test exhibited a significant difference with $F_{(2,76)}$ =7.074, P=0.029 and $F_{(3,20)}$ =10.872 P=0.012 for Elgeyo and Nyambene, respectively. The estimated proportion of households visiting the two ecosystems for recreational purposes is approximately 7% and 5% of the total for Elgeyo and Nyambene, respectively. Likewise, the average number of day visits for recreational services is about 26.63±21.34 and 30.35±26.86 for Elgeyo and Nyambene, respectively. Using the cultural mean WTP of USD 7.4 per household, the study estimates the recreational value at KES 199.3 million (USD 1.86 million) and KES 207.7 million (USD 1.94 million) for the Elgeyo and the Nyambene ecosystems, respectively (Table 4.21). This translates to KES 1,843.93 (USD 17.23) and KES 6,894.24 (USD 64.43) $ha^{-1}year^{-1}$ for

Elgeyo and Nyambene, respectively. The recreational unit values were higher compared with a study in Chile National System of Public Protected Areas (SNASPE), which value recreation at USD 6.3 $ha^{-1}year^{-1}$ (Puyehue ecosystem) at a study at Vicente Pérez Rosales ecosystem which was valued at USD1.6 $ha^{-1}year^{-1}$ (Nahuelhual et al., 2007). Variances in ecosystem size, number of visits, and the methodology used to assign monetary value could explain the differences in cultural and recreational values.

Table4.21: Cultural/Spiritual and Recreational Services Values (HH Annual) Statistics

Ecosystem	**ES**	**Valid (%)**	**WTP (USD)**	**Visitation Frequency** Min	Max	Mean	**Std. Dev**	**Value (KES) million**	**Value (USD) million**
Elgeyo	Cultural	20.38	7.40	1.00	48.00	6.08	13.72	128.08	1.20
	Recreation	7.24	7.40	1.00	48.00	26.63	21.34	199.33	1.86
Nyambene	Cultural	53.98	7.40	1.00	240.0	32.78	36.49	2,434.5	22.75
	Recreation	4.98	7.40	2.00	96.00	30.35	26.86	207.72	1.94

4.2.1.6 Summary of Direct Use ES Values (DUV)

The study estimates the aggregate economic value for direct use (DUV) forest products at KES 12.8 billion (USD 119.3 million) and KES 11.6 billion (USD 108.8 million) for the Elgeyo and the Nyambene ecosystems, respectively. The findings correspond to a unit value of KES 97,749.32 (USD 913.55) and KES 66,994.90 (USD 626.12) $HH^{-1}year^{-1}$ for the Elgeyo and the Nyambene ecosystems, respectively. Equally, this equates to KES 118,092.21 (USD1,103.67) and KES 386,322.42 (USD3,610.49) ha^{-1} $year^{-1}$ (Table 4.23). The study estimates were higher compared to a study of Mau complex Cherangany and Mt Elgon, which estimated DUV at KES 32,862.88 (USD 307.13) $ha^{-1}year^{-1}$ (Langat et al., 2021). Equally, higher compared to a study in East Mau, Transmara, and Masai Mau forest, which estimated DUV at KES 15,266.6 (USD 146.02) $ha^{-1}year^{-1}$ (Kipkoech et al., 2011) However, the study estimates were lower compared to a study in a tropical forest in Malaysia, which estimated the DUV using timber stumpage value at RM257,075 (USD 56,624.45) $ha^{-1}year^{-1}$ (Nitanan et al., 2020). The study attributes the difference to the number of beneficiaries, level of accessibility, conservation status, and the number of DUV ES incorporated and valuation methods.

Table 4.22: Total Annual Household Direct Use Products and Economic Value

ES	Elgeyo			Nyambene		
	Mean	Value (KES) million	(%)	Mean	Value (KES) million	(%)
Fuelwood	37,902.44	3,317.66	25.99	26,059.64	3,446.44	29.61
Timber	42,000.00	14.71	0.12	13,766.67	89.25	0.77
Charcoal	183,788.89	1,737.43	13.61	25,221.82	359.73	3.09
Honey	64,500.00	180.67	1.42	33,615.38	377.74	3.25
Natural Medicine	7,845.00	32.96	0.26	12,600.00	43.57	0.37
Fencing Poles	82,060.00	430.97	3.38	22,650.00	19.58	0.17
Building Poles	50,000.00	140.05	1.10	750.00	0.65	0.01
Wild Fruits	2,615.00	4.58	0.04	472.22	7.35	0.06
Fodder	30,035.14	73.61	0.58	56,172.21	1,262.43	10.85
Grazing Value	136,330.00	3,418.42	26.78	77,445.60	1,076.45	9.25
Farming Tools	175.00	0.25	0.00	55.77	1.25	0.01
Quarry Stones	-	-	-	140,000.00	181.52	1.56
Marram	133,333.33	140.05	1.10	-	-	-
Mushroom	10,862.31	49.44	0.39	99,900.00	86.35	0.74
Reeds	-	-	-	60,000.00	25.93	0.22
Game Meat	-	-	-	21,600.00	18.67	0.16
Domestic Water	7,668.38	1,001.47	7.84	7,477.78	1,299.21	11.16
Livestock Water	15,175.64	1,902.22	14.90	5,095.03	750.94	6.45
Cultural/Spiritual Values	4,723.34	125.69	0.98	25,472.71	2,389.00	20.52
Recreational Values	20,691.22	195.60	1.53	23,581.95	203.84	1.75
Total		12,765.77	100.0		11,639.89	100.0

4.2.2 Indirect Use Values

The section presents findings on indirect use values, grouped as regulatory and support services in the millennium ecosystem assessment typology. This includes water flow regulation, water purification, soil conservation, nutrient conservation, climate regulation, and the influence on microclimate and crop pollination, among others. As aforementioned, the valuation of these services is complex since most of the products and benefits are intangible and not traded in conventional markets, thus no unit price exists. In the unit's price absence, the study employed the benefits transfer technique.

4.2.2.1 Water flow regulation

Based on the mean annual precipitation of 1200 mm (Elgeyo) and 1400mm (Nyambene) and respective runoff coefficients, the two ecosystem preserves about 70 million and 39 million cubic meters of precipitation water annually translating. Using a unit cost of USD 2.1 per m^3 for constructing and maintaining an artificial water storage dam as a surrogate the study values watershed protection at KES 15.6 billion (USD146.2 million) and KES 8.6 billion (USD81 million) for the Elgeyo and Nyambene ecosystems, respectively. The aggregated estimates translate to KES 620,200/ (USD 5,796.3) and KES 1,600,000/ (USD 14,953.30) per hectare annually, respectively (Table 4.23). The study estimates were higher compared to a study in the Mau East ecosystem valued at KES 127, 893.11 (USD 1,421.03) $ha^{-1}yr^{-1}$ (Langat, 2016) and a study in Indonesia that reported water flow regulatory and maintenance services at USD 1880 $ha^{-1}yr^{-1}$ (range of USD 707–3110 $ha^{-1}yr^{-1}$) (Aulia et al., 2020). Similarly, the study estimates were higher than the study in China, which placed a value between USD 540 and 560 per ha annually (Xi, 2009). It attributes the variance to the difference in runoff coefficients, mean annual rainfall, forest cover, and the unit cost for the surrogate reservoir which vary across ecosystems and jurisdictions. Equally, some studies considered the reservoir establishment cost but didn't factor in the operation and management cost of the reservoir.

Table 4.23: Watershed Protection Valuation

Land Cover/Use	Elgeyo		Nyambene	
	Water Conserved (m3)	**Water Conservation Value (KES)**	**Water Conserved**	**Water Conservation Value (KES)**
Dense Forest	34,566,564.2	7,685,875,545.64	35,813,186.8	7,963,062,092.16
Moderate Forest	10,142,986.9	2,255,293,133.48	1,776,321.47	394,965,079.31
W/Grassland	6,013,916.68	1,337,194,373.12	379,161.17	84,306,486.60
Bushland	3,486,764.97	775,282,191.97	-	-
P/Cropland	772,745.02	171,819,856.02	729,679.64	162,244,267.23
A/Cropland	15,085,939.1	3,354,358,562.81	281,924.17	62,685,838.91
Wetland	225,504.49	50,140,922.61	-	-
Open Water	67,553.13	15,020,439.23	-	-
Other Land	-	-	16,963.24	3,771,776.95
Fallow land	-	-	-	-
Total	70,361,974.5	15,644,985,024.89	38,997,236.5	8,671,035,541.16

4.2.2.2 Water Quality Regulation

Based on the preservation principle, the two ecosystems potentially preserve precipitation estimated at 70 million m^3 and 38 million m^3 for the Elgeyo and the Nyambene, respectively. Using the surrogate (municipal wastewater treatment plant) unit cost of USD 0.3/m^3, the study estimated the economic value for the two-ecosystem water purification service at KES 2.2 billion (USD 20.6 million) and KES 1.2 billion (USD 11.2 million). This translates to KES 87,862.3 (USD 821.14) per ha year and KES 226,340.86 (USD 2,115.34) per ha annually respectively (Table 4.24). These estimates were higher compared to a study in East Mau, which estimated water purification function at about KES 1,000 (USD 9.35) $ha^{-1}year^{-1}$ (Langat, 2016). Equally higher than a study in Mau complex, Cherangany, and Mt Elgon, which valued water quality function at KES 897.92 (USD 8.39) $ha^{-1}year^{-1}$ (Langat et al., 2021). However, they were within the range for the case of the Elgeyo ecosystem though higher for the case of Nyambene ecosystems when compared with a study in China that valued forest water purification function at between USD 999.55 and USD 1,105.70 $ha^{-1}year^{-1}$ (Xi, 2009). The study attributes the discrepancies to the data used in the different studies and the variance in the runoff coefficient. For instance, the East Mau study used data on domestic water, while the study employed the potential precipitation water preserved by the ecosystem.

Table 4.24: Ecosystem Water Purification Function Valuation

Land cover	Elgeyo		Nyambene	
	Water (m^3)	**Value (KES)**	**Water (m3)**	**Value (KES)**
Dense Forest	34,566,564.18	1,088,846,771.7	35,813,186.8	1,128,115,385.22
Moderate Forest	10,142,986.88	319,504,086.82	1,776,321.47	55,954,126.37
W/ Grassland	6,013,916.68	189,438,375.32	379,161.17	11,943,576.92
Open Grassland	3,486,764.97	109,833,096.68	-	-
P/Cropland	772,745.02	24,341,468.25	729,679.64	22,984,908.56
A/Cropland	15,085,939.12	475,207,082.21	281,924.17	8,880,611.31
Wetland	225,504.49	7,103,391.33	-	-
Open Water	67,553.13	2,127,923.70	-	-
Other lands	-	-	16,963.24	534,342.14
Fallow land	-	-	-	-
Total	70,361,974.48	2,216,402,196.0	38,997,236.5	1,228,412,950.51

4.2.2.3 Soil Conservation

The study assessed the forest soil conservation function based on the amount of soil lost through soil erosion. It assumes that a proportion of soil lost through erosion finds its way into water bodies as sediments. The study estimates the forest soil conservation for the two ecosystems at 1.34 million tons and 0.4 million tons of soil for the Elgeyo and the Nyambene respectively annually. In that regard, using the soil *ex-situ* sedimentation and dredging unit cost of USD 3.34 as a proxy, the study estimates soil conservation value at KES 478 million (USD 4.4 million) and 130.7 million (USD 1.22 million) respectively (Table 4.25). Estimates translate to KES 18,943.62 (USD 177.04) and 24,081.21 (USD 225.06) per ha per year, respectively. The findings were higher compared with a study in Mau, Cherangany, and Elgon, which value soil conservation at KES 823.80 (USD 7.6) ha^{-1} $year^{-1}$ (Langat et al., 2021). However, they fall within the range in comparison with the Shangyon-Mengla study, in China, which ranged between USD 49.86 and 1,096.39 ha^{-1} $year^{-1}$ (Xi, 2009). But higher compared to a study in Anji County, Huzhou, Zhejiang, China, which valued soil conservation at 436 RMB (USD 69.8) ha^{-1} $year^{-1}$ (B. Zhang et al., 2015). The study attributed the inconsistency to the land cover proportion and respective coefficients, the sediment ratio, and proxy unit cost. The study considers that all the soil lost found its way to the water reservoir. However, the study in Mau, Cherangany, and Elgon assumed that only 50% of the sediments get into the water bodies.

Table 4.25: Economic Valuation of Forest Soil Conservation

Land Cover	Elgeyo		Nyambene	
	Total Conserved	**Economic Value**	**Total Soil Conserved**	**Economic Value2**
Dense Forest	556,511.18	195,335,422.91	329,475.10	115,645,759.53
Moderate Forest	163,299.01	57,317,951.01	16,341.85	5,735,989.00
W/Grassland	226,658.66	79,557,188.16	-	-
Bushland	137,477.98	48,254,769.92	6,530.64	2,292,254.02
Crop Land	244,924.33	85,968,439.59	-	-
P/Crop land	27,779.82	9,750,717.37	14,989.51	5,261,317.73
Wetland	3,689.79	1,295,114.71	4,991.51	1,752,021.60
Other Land	-	-	-	-
Open Water	1,099.70	385,994.70	253.29	-
Fallow land	-	-	147.49	-
Total	1,361,440.45	477,865,598.37	372,729.39	130,687,341.87

4.2.2.4 Soil Nutrient Conservation

The study assessed and recorded soil minerals in Elgeyo, as shown in (Table 4.26). The soil moisture content had a mean ± std. deviation of 57.47±8.46%. Other minerals recorded a mean± std deviation of 7.86±0.8% (soil organic carbon), 13.56±1.38 (soil organic matter), 0.91±0.1 (organic Nitrogen), 12.65±10.9 (Phosphorus), 0.91±0.11 (soil bulk density) and 5.68±0,37 (pH). Soil mineral analysis across the Elgeyo land cover/use exhibited non-significance, suggesting that the difference in land cover with the ecosystem doesn't influence soil mineral composition. The levels established in the study were higher compared to a study in Kaaga Meru County Kenya that established nitrogen, potassium, and phosphorus at 0.17±0.09%, 0.85±0.11mg/Kg, and 2.26±0.04 mg/Kg, respectively (Kianira, 2021). Equally higher compared to a study in selected farms in Kakamega County, Kenya where both soil nutrients were below the critical levels, that is nitrogen 0.15% (CL= 0.2%), potassium 1.20±0.01 mg/kg (CL= 2.4 mg/kg) (Lumula, 2021). The study pH falls within the acceptable critical range of 5.5 and 6.5 (NAAIAP & KARI, 2014)

Table 4.26: Forest Soil Mineral Statistics (Elgeyo)

Land Cover Category	**MC (%)**	**pH**	**EC (mS/cm)**	**% N**	**Olsen P (ppm)**	**SOC (%)**	**SOM (%)**	**BD (g/cm^3)**
Dense forest	60.33	5.85	0.18	0.89	11.86	7.77	13.39	0.90
Moderate	51.88	5.64	0.25	0.96	12.78	8.29	14.29	0.85
W/grass land	58.32	5.64	0.28	0.91	11.84	7.75	13.36	0.90
Bushland	63.27	5.76	0.21	0.91	12.78	7.80	13.44	0.94
Degraded	57.58	5.66	0.28	0.91	14.17	7.88	13.59	0.91
PELIS	55.35	5.46	0.23	0.94	8.34	7.89	13.61	0.93
Glades	56.99	5.72	0.25	0.89	14.57	7.69	13.26	0.94
Total	58±8	5.7±0.4	0.3±0.2	0.9±0.1	12.7±10.9	7.9±0.8	13.6±1.4	0.9±0.2

The soil mineral assessment for the Nyambene ecosystem recorded soil moisture content with a mean± std. dev of 14.12±4.18 percent. the study also recorded other minerals with a mean± std dev of 4.19±1.58% (soil organic carbon), 7.23±2.73 (soil organic matter), 0.85±0.25% (Nitrogen), 6.07±1.92ppm (Phosphorus), 388.45±239.3 ppm (Potassium) 1.21±0.20 (soil bulk density) and 4.84±0.471 (pH) (Table 4.27). The soil nutrient levels were high compared to studies in Kaaga Meru (Kianira, 2021) and selected farms in Kakamega (Lumula, 2021), most of which recorded levels below critical values. Soil mineral analysis across the Nyambene land cover/use exhibited significant difference for soil carbon, organic matter, and nitrogen. However, the other minerals exhibited non-significance. All the soil minerals assessed recorded values above the critical levels and within the acceptable levels (NAAIAP & KARI, 2014)

Table 4.27: Forest Soil Mineral Statistics (Nyambene)

Nyambene Land Cover	pH	EC (mS/cm)	MC (%)	BD (g/cm3)	SOC (%)	OM (%)	N (%)	P (ppm)	K (ppm)
Dense forest	4.61	0.12	15.51	1.18	4.94	8.52	0.86	6.23	337.93
Moderate	4.71	0.09	13.73	1.18	4.58	7.90	0.94	6.10	403.56
Open forest	5.15	0.07	14.51	1.20	4.16	7.17	0.91	6.07	479.23
Other lands	5.06	0.15	11.77	1.42	1.60	2.76	0.48	5.72	262.56
Total	4.8±0.5	0.10±0.1	14.1±4.2	1.21±0.2	4.19±1.6**	7.23±2.7**	0.85±0.3*	6.08±1.9	388.45±239.3

Using the mean soil loss per hectare (erosion rate) with the proportionate soil nutrient across the land cover, the study estimates the amount of nutrients preserved by the two ecosystems at 12,277.54 mg and 3,219.13 $mg^{-1}year^{-1}$ for Elgeyo and Nyambene, respectively. Using the unit cost of KES 60 per kg of commercial fertilisers as a proxy, the study value for soil nutrient preservation function at KES 3.6 billion (USD33.3 million) and KES 935.4 million (USD8.74) annually for the Elgeyo and Nyambene ecosystems, respectively. This, respectively, corresponds to KES 141,519.36 (USD 1,322.61) and KES 172,442.02 (USD 1,611.61) per hectare per year (Table 4.28). Study results were higher compared with the Mau, Cherangany, and Elgon water towers, that estimated soil nutrient conservation at KES 3,496.49 (USD 32.68) ha^{-1} yr^{-1}, respectively (MoE & F, 2019a). Equally, higher than a study in Chile soil fertility conservation valued at USD 26.3 ha^{-1} yr^{-1} (Nahuelhual et al., 2007), Anji County, Huzhou, Zhejiang, China, on forest soil conservation based on the eco-service unit method with a mean value of RMB 436 (USD 69.8) $ha^{-1}yr^{-1}$ (B. Zhang et al., 2015). However, the estimates were slightly higher though within the range of a study in the Xishuangbanna corridor in China with a mean value of USD 1,103.61 $ha^{-1}yr^{-1}$ (Xi, 2009).

Table 4.28: Forest Soil Nutrient Conservation Value

Land Cover	Elgeyo		Nyambene	
	Soil Nutrient Conserved (Mg)	**Economic Value (KES)**	**Soil Nutrient Conserved (Mg)**	**Economic Value (KES)**
Dense Forest	5,063.48	1,471,388,000.64	2,950.88	857,490,379.30
Dense Exotic Forest	1,435.25	417,067,439.23	151.20	43,936,572.24
Wooded Grassland	2,057.72	597,950,303.93	59.53	17,297,375.02
Bushland	1,248.09	362,680,647.54	-	-
Crop Land	2,181.49	633,915,864.13	34.33	9,975,574.04
Perennial Cropland	247.43	71,899,758.49	22.45	6,524,256.03
Vegetated Wetland	34.47	10,017,201.26	-	-
Other Land	-	-	-	-
Open Water	-	-	-	-
Fallow land	9.60	2,790,441.41	0.75	217,338.70
Total	12,277.54	3,567,709,656.63	3,219.13	935,441,495.34

4.2.2.5 Plant Carbon Sequestration

The study estimated plant carbon using an improved allometric regression model (Chave et al., 2014). It later classified the estimates based on the IPCC land cover and land use categories, namely dense forest, moderate and wooded grassland, bushland, perennial cropland, cropland, and vegetated wetland. The mean tree carbon for the Elgeyo varied significantly across the quoted land cover/ land use with $F_{(7,47)}$ =4.389, $P<0.05$, with an overall Mean ± Std dev. 56.05±67.03 Mg Carbon /ha and ranges from 0 to 144.67±86.37Mg/ha for cropland and dense forest, respectively. The Nyambene as well varied significantly across the land cover with $F_{(4,26)}$ =7.205, $P<0.01$ ranging between 0 to 210.47±100.41Mg/ha for cropland and dense forest respectively and with an overall Mean ± std deviation of 130.02±103.68Mg/ha (Table 4.29). The study estimates were higher than most dry land forest carbon, including studies in Cameroon (Kemeuze et al., 2015), a study in Miombo woodland (Lupala et al., 2014), and a Marsabit ecosystem (Muhati et al., 2018). However, they were lower compared to a study on carbon storage across global tropical forests (Sullivan et al., 2017), African tropical forests (Lewis et al., 2013), and Borneo tropical forests (Slik et al., 2010). That notwithstanding, the study estimates were within ranges compared to a study in the

Taita Hills (Omoro et al., 2013). The study could attribute perhaps this to degradation, deforestation, and conversion of the forest to other land uses (Domke et al., 2019; KWTA, 2020b).

Table 4.29: Forest Dimension Statistics (Average unit)

Land Cover type	Elgeyo				Nyambene			
	Trees/ha	DBH (cm)	Height (m)	Carbon (Mg/ha)	Trees/ha	DBH	Height (m)	Carbon (Mg/ha)
Dense forest	888	20.7	17.7	144.67	700	66.16	14.13	210.47
Moderate forest	548	22.0	17.4	80.14	667	47.69	12.93	151.34
W/grass land	269	24.9	16.1	82.16	524	52.97	15.90	84.47
Bushland	73	29.6	17.0	25.15	118	91.43	15.44	24.07
Cropland	-			-				
Open forest	79	32.1	20.6	36.80	290	52.60	12.91	20.29
Others Land	169.9	6.16	5.57	0.43				
Total	258	25.1	17.0	56.06	523	64.12	14.00	130.02

Overall, the study estimates that the two ecosystem store approximately 1.7 million Mg of C and 1.1 million Mg of C of carbon for Elgeyo and Nyambene, respectively. This translates to reduced emissions of about 6.3 million for Elgeyo and 3.9 million tons of CO_2 equivalent for Nyambene. Using a unit price of USD 5 per ton of CO_2, the study values the forest carbon of the two ecosystems at KES 3.4 billion (USD 31.4 million) and KES 2.1 billion (USD 19.3 million) respectively. The estimates translate to KES 133,203.57 (USD1,244.89) and KES 380,189.55 (USD 3,553.10) per ha annually, respectively (Table 4.30). The estimates were lower for the case of the Elgeyo and higher for the case of Nyambene in comparison with the Mengla-Shangyon and Nabanhe-Mangao corridor (China) study with a unit value of USD 2,195 per ha annually (Xi, 2009). A similar scenario compared to the East Mau study that valued carbon sequestration at USD 2,782.47 per ha annually (Langat et al., 2016). The study mainly attributed the difference to the carbon unit price, whereby the two reference studies used USD 10 while this study used USD 5 per unit of CO_2. Similarly, most of the reference studies used unit transfer with an assumption that the mean carbon estimates are similar across different land cover/uses. Notably, the study confirms that forest carbon varies significantly across ecosystems and landscapes.

Table 4.30: Forest Carbon Valuation

Land Cover	Elgeyo		Nyambene	
	Carbon (Mg)	**Total Value (USD)**	**Carbon (Mg)**	**Total Value (USD)**
Dense forest	1,166,781.84	21,410,446.69	1,004,970.76	18,441,213.37
Moderately forest	189,670.65	3,480,456.34	35,843.37	657,725.92
W/grass land	286,493.72	5,257,159.77	8,112.63	148,866.72
Bushland	50,847.46	933,050.91	-	-
Open forest	16,487.04	302,537.25	1,449.10	26,590.89
Others such as PELIS	9.03	165.75	-	-
Glades	-	-	-	-
Total	1,710,289.74	31,383,816.70	1,050,375.85	19,274,396.91
Unit Value	67.84	1,244.89	193.63	3,553.10

4.2.2.6 Soil Carbon Sequestration

The soil carbon assessment in Elgeyo exhibited non-significant differences between land covers, with a mean ± standard deviation of 213.71±43.23 Mg SOC/ha. This amount is much higher than forest tree carbon, demonstrating the significant potential of soils in this ecosystem to store significant amounts of soil carbon stock. The Elgeyo findings suggest that the difference in land use doesn't have a significant influence on soil organic carbon stock and economic value consistent with a study in Miombo woodland in Mozambique (Williams et al., 2007). However, inconsistent with a study in the Mau ecosystem estimated at between 116.5 ± 40 and 135.2 ± 360 Mg/ha (Tarus & Nadir, 2020). The difference in the estimates would be attributed to the higher bulk density compared with the Mau ecosystem study that recorded between 0.59 g/cm3 and 0.66 g/cm^3. However, the Nyambene ecosystem recorded significant differences across the land cover with $F_{(3,20)} = 3.986$, $p<0.05$ ranging from 67.81±37.71 to 170.61 ±68.06 with a mean of 133±50.87 Mg SOC/ha Equally, the Nyambene soil carbon was consistent with a study in the Mau ecosystem estimated at between 116.5 ± 40 and 135.2 ± 360 Mg/ha (Tarus & Nadir, 2020). Overall, both the ecosystems' SOC estimates fall within range with a study in Taita Hills, estimated between 78 and 305 Mg C/ha from 0 to 50 cm layer (Omoro et al., 2013).

Based on the respective mean SOC, the study estimates the aggregate soil carbon for the two ecosystems at 8.2 million and 0.87 million Mg SOC. This is equivalent to 30.2 million and 3.2 million mg of CO_2e for Elgeyo and Nyambene, respectively. At a unit price of USD 5, the total value for soil CO_2e sequestered by the two ecosystems is KES 16.1 billion (USD 150.85 million) and 1.7 billion (USD 15.95 million) respectively. This corresponds to KES 639,866.42 (USD 5,980.06) and KES 314,662.39 (USD 2,940.77) per hectare per year (Table 4.31).

Table 4.31: Forest Soil Carbon (SC) Valuation

Land Cover	Elgeyo			Nyambene		
	SOC (mg/ha)	Total SOC (Mg)	Total Value (USD)	SOC (mg/ha)	Total SOC (Mg)	Total Value (USD)
Dense forest	209.33	1,688,328.0	30,980,818.70	170.61	814,663.0	14,949,065.93
Moderate	212.57	1,846,290.0	33,879,421.93	152.00	35,999.44	660,589.64
W/grassland	208.77	2,671,679.3	49,025,315.63	141.69	13,607.74	249,701.98
Bushland	220.21	1,633,869.1	29,981,497.19		-	-
Cropland		-	-		-	-
Degrade	213.52	351,113.79	6,442,937.97	67.81	4,842.27	88,855.64
Other lands	218.76	16,924.75	310,569.08			
Glades	217.50	12,622.21	231,617.59		-	-
Total	213.71	8,220,827.14	150,852,178.09	133.03	869,112.43	15,948,213.18
Unit Value		325.89	5,980.06		160.21	2,939.94

The study estimates the aggregate carbon sequestered by the two ecosystems at 9.9 million and 1.92 million tons, translating to 36.5 million and 7.04 million tons of CO_2e for the Elgeyo and Nyambene, respectively. This translates to aggregate estimates of KES 19.5 billion (USD 182.5 million) and KES 3.77 billion (USD 35.2 million) for the Elgeyo and Nyambene ecosystems, respectively.

4.2.2.7 Oxygen generation Valuation

The ratio of CO_2 to oxygen from the photosynthesis formula is 1: 0.73, which translates to one unit of CO_2 would be equivalent to 0.73 units of oxygen. In this regard, the estimated value of the oxygen generated by the two ecosystems is 13 million m^3 and 8 million m^3 for the Elgeyo and the Nyambene ecosystems, respectively. Based on the LOX unit price, the study estimates the value for the two ecosystems at KES 3.2 billion (USD 29.8 million) and 1.96 billion (USD 18.3 million) for Elgeyo and Nyambene, respectively (Table 4.32). This translates to KES 126,850.83 (USD1,185.52) and 361,291.85 (USD3,376.56) per ha annually, respectively. The estimates are within range compared with the values of study in Mau, Elgon, and Cherangany ranging between 1,275.29 and 5,476.18 per ha annually (Langat et al., 2020). However, the findings were lower compared to the Xishuangbanna biodiversity conservation corridor in China, ranging between USD 8,000 and 10, 000 per ha annually (Xi, 2009). The study would attribute this to the surrogate unit cost, where, for instance, Mau, Elgon, and Cherangany used USD 5 per unit cost of surrogate while the China study used USD 6.5 per unit.

Table 4.32: Oxygen Generation Valuation

Land Cover	Elgeyo			Nyambene		
	Tree CO_2 (mg/ha)	Total Oxygen generated (M^3)	Oxygen generation Value (USD)	Tree CO_2 (mg/ha)	Total O_2 generated (M^3)	O_2 Generation Value (USD)
Dense forest	530.92	8,846,368.4	20,346,647.2	772.41	7,619,540.54	17,524,943.3
Moderate forest	294.13	1,438,055.0	3,307,526.39	555.42	271,759.20	625,046.15
W/grassland	301.52	2,172,153.3	4,995,952.53	310.01	61,508.75	141,470.13
Bushland	92.30	385,518.0	886,691.34	88.34	-	-
Degraded forest	-	125,002.3	287,505.38	-	10,986.83	25,269.70
PELIS Land	135.04	68.48	157.51	74.47	-	-
Total	1.57	12,967,165.4	29,824,480.4	-	7,963,795.3	18,316,729.2

NB: 1mg is equivalent to 2.832 m^3

4.2.2.8 Microclimate Function

The study used the household cash crop data yields supported information generated from the government (county and national) officials. This was extrapolated into annual estimates as summarised in (Table 4.33) Based on the average band of 15% influence on cash crop production, the study estimates the economic value for microclimate at KES 0.45 billion (USD4.3 million) and KES 6.7 billion (USD63.4 million) for the Elgeyo and the Nyambene, respectively. The estimates translate to KES 17,959.29 (USD 171.04) and KES 1,254,897.59 (USD 11,951.41) per ha annually, respectively. The study estimates are high compared to the findings of a study in Mau, Cherangany, and Elgon that value microclimate function at KES 3,262.82 (USD 30.49) $ha^{-1}year^{-1}$ (Langat et al., 2021; MoE & F, 2019a). Equally higher than a study in Transmara, Maasai Mau, and East Mau that estimated microclimate at KES 1,736.30 (USD 16.23) $ha^{-1}year^{-1}$ (Kipkoech et al., 2011). The difference is attributed to the number of crops incorporated, where the reference study only incorporated tea production as a proxy, while the study included a wide range of crops, as stated in the table (Table 4.33). Worth noting the estimates mainly considered cash crop production value with an assumption that the food crop values were conservative and thus not incorporated into the valuation of the microclimate services.

Table 4.33: Microclimate Influence Valuation

Cash Crops	Elgeyo			
	Total Annual (kg)	**Total Crop Value (KES)**	**Microclimate Value (KES)**	**Microclimate Value (USD)**
Coffee	46,220.63	8,654,812.03	2,163,703.01	20,606.70
Cotton	1,485,000.00	317,790,000.00	47,668,500.00	453,985.71
Pyrethrum	1,113,750.00	445,500,000.00	66,825,000.00	636,428.57
Sunflower	445,500.00	13,921,875.00	2,088,281.25	19,888.39
Tea	5,801,400.00	1,169,562,240.00	175,434,336.00	1,670,803.20
Wheat	32,076,000.00	1,058,508,000.00	158,776,200.00	1,512,154.29
Sub Total	-	3,013,936,927.03	452,956,020.26	4,313,866.86
Cash crop	Nyambene			
Coffee	38,096,434.76	7,133,557,407.92	1,070,033,611.19	10,190,796.30
Cotton	90,346.36	19,334,121.04	2,900,118.16	27,620.17
Macadamia	2,731,997.13	3,958,800,444.12	593,820,066.62	5,655,429.21
C. edulis	10,999,998.05	11,549,997,954.34	1,732,499,693.15	16,499,997.08
Tea	107,774,867.82	21,727,413,351.71	3,259,112,002.76	31,039,161.93
Sub Total		44,389,103,279.13	6,658,365,491.87	63,413,004.68

(Crop Data Source: Interviews of households and county government officials)

4.2.2.9 Pollination

The study used the household average yield data to estimate the respective crop annual production. This was supported by data generated from the county (Elgeyo Marakwet, Uasin Gishu, and Meru) and national government databases (Appendix V). Using the pollination dependency function (FAO, 2005), the study estimates the economic value of natural pollinators contributed to crop production around the two ecosystems at KES 821.5 million (USD 7.82million) and 4.06 billion (USD 38.68 million) for Elgeyo and Nyambene ecosystems, respectively (Table 4.34). This translates to KES 32,633.17 (USD 310.79) and KES 765,429.51 (USD 7,289.80) per ha annually, respectively. The Estimates represent approximately 2% and 8% of the total market value for annual crop production for the household around the Elgeyo and Nyambene ecosystems, respectively. The vulnerability rate of 2% and 8% respectively regarded as vulnerability rate, is lower for the case of Elgeyo, but consistent for the case of the Nyambene ecosystem with a global average vulnerability rate of 9.5% (Gallai et al., 2009).

Table 4.34: Insect Pollination Valuation

Crop	Elgeyo		Nyambene	
	Total Crop Value (KES)	**Insect Pollination Value (KES)**	**Total Crop Value (KES)**	**Insect Pollination Value (KES)**
Avocado	97,315,393.75	63,255,005.94	1,791,351,914.94	1,164,378,744.71
Bananas	127,940,958.75	-	4,252,696.99	-
Beans	156,395,250.00	39,098,812.50	1,385,179,772.70	346,294,943.17
Cabbages	1,389,417,486.03	-	113,005,743.45	-
Capsicum	-	-	546,595.48	27,329.77
Carrot	6,424,687.50	-	63,901,372.33	-
Citrus	2,812,813.37	1,828,328.69	174,474,402.00	113,408,361.30
Coffee	8,654,812.03	2,163,703.01	7,133,557,407.92	1,783,389,351.98
Cotton	317,790,000.00	79,447,500.00	19,334,121.04	4,833,530.26
Cowpeas	-	-	27,582,232.21	6,895,558.05
Dolichos	-	-	265,227,506.96	66,306,876.74
French bean	-	-	7,981,319.06	399,065.95
G/grams	674,364,269.21	168,591,067.3	8,797,476.81	2,199,369.20
Green Peas	105,851,625.00	26,462,906.25	181596.18	45,399.05
G/pepper	-	-	198,364,219.56	9,918,210.98
Guavas	-	-	106,365,096.26	26,591,274.07
I/Potatoes	7,504,294,448.84	375,214,722.4	575,372,392.10	28,768,619.60
Maize	31,252,418,634.1	-	7,125,181,422.52	-
Mangoes	1,961,697.48	1,275,103.36	60,978,867.39	39,636,263.80
Melon	1,081,512.61	270,378.15	-	-
Millet	594,742,500.00	-	-	-
C. edulis	-	-	11,549,997,954.3	-
Onions	458,132,567.80	22,906,628.39	222,152,777.54	11,107,638.88
Pawpaw	1,020,334.26	255,083.57	8,061,327.71	2,015,331.93
Pigeon peas	2,613,600.00	653,400.00	3,975,239.84	993,809.96
Pumpkins	24,066,298.34	-	61,839,442.53	-
Pyrethrum	445,500,000.00	-	-	-
Red Pepper	-	-	6,026,175.01	301,308.75
Soya beans	-	-	3,201,649.13	800,412.28
Sunflower	13,921,875.00	3,480,468.75	-	-
Tea	1,169,562,240.00	23,391,244.80	21,727,413,351.7	434,548,267.03
Tomatoes	263,653,109.24	13,182,655.46	368,655,214.07	18,432,760.70
Wheat	1,058,508,000.00	-	-	-
Total	**45,678,444,113.3**	**821,477,008.6**	**53,012,959,287.8**	**4,061,292,428.18**

(Source of crop data: interviews of household and county government officials; and insect pollination dependency factor sourced from FAO, (2005), database).

4.2.2.10 Summary Indirect Use Values (IUVs

The study estimates the aggregates economic values for regulatory ecosystem services (RES) at KES 45.8 billion (USD 428.5 million) and KES 27.4 billion (USD 256.2 million) for the Elgeyo and the Nyambene ecosystem, respectively. This translates to KES 423,728.07 (USD 3,960.08) and KES 907,746.51 (USD 8,483.61) per ha annually, respectively. Elgeyo ecosystem, carbon sequestration, and watershed protection account for over 70% of indirect use values. While the Nyambene ecosystem, watershed protection, microclimate influence, pollination, and carbon sequestration account for over 80% of IUVs (Table 4.35). The findings were high compared to a study in Mau complex, Cherangany, and Mt Elgon that valued IUV at KES 239,238.50 (USD 2,235.87) $ha^{-1}year^{-1}$ (Langat et al., 2021; MoE & F, 2019a) and a study in Malaysia, which estimated IUVs at RM 9,757.05 (USD 2,144.41) $ha^{-1}year^{-1}$ (Nitanan et al., 2020). It was lower for the case of Elgeyo and higher for the case of Nyambene compared to a study in East Mau that valued IUVs at KES 641,212.75 (USD 5,992.64) $ha^{-1}year^{-1}$ (Langat, 2016). However, the study estimates were lower compared with the study in Nabanhe-Mangao and Mengla-Shangyong corridors valued at USD 12,947.71 and USD 389,248.33 ha^{-1} $year^{-1}$ respectively (Xi, 2009). The study would attribute the difference in terms of the incorporated ES and proxy unit prices. Notably, valuation is very sensitive to unit prices, that is any change and/or choice of a product or surrogate unit price is likely to either increase or reduce the economic values.

Table 4.35: Summary of Indirect Use Values

Ecosystem Services	Elgeyo		Nyambene	
	Value (KES)	%	Value (KES)	%
Watershed protection	15,644,985,024.48	34.12	8,671,035,541.16	31.63
Water purification	2,216,402,196.02	4.83	1,228,412,950.51	4.48
Soil Conservation	477,865,598.37	1.04	130,687,341.87	0.48
Soil Nutrient	3,568,003,646.75	7.78	935,441,495.34	3.41
Plant Carbon	3,358,068,386.79	7.32	2,062,360,469.25	7.52
Soil Carbon	16,116,399,974.10	35.15	1,706,458,810.28	6.22
Oxygen generation	3,191,219,400.92	6.96	1,959,890,026.97	7.15
Microclimate	452,956,020.26	0.99	6,658,365,491.87	24.29
Pollination	821,477,008.61	1.79	4,061,292,428.18	14.81
Total	45,847,377,256.31	100.00	27,413,944,555.43	100.00

4.2.3 Non-use Values

Using the mean willingness to pay (WTP) in the Elgeyo for passive ecosystem services by Eregae et al. (2021), the study estimates the bequest value at KES 93.6 million (USD0.9 million) and KES 164.1 million (USD1.5 million) for Elgeyo and Nyambene, respectively. The existence value at KES 114.2 million (USD1.1 million) and KES 200.2 million (USD1.9 million), respectively (Table). Overall, the study estimates the aggregate values for non-use values at KES 207.8 million (USD 1.9 million) and KES 364.3 million (USD 3.4 million) for Elgeyo and Nyambene, respectively (Table). This translates to KES 1,922.37 (USD 17.97) and KES 12,021.86 (USD 112.35) $ha^{-1}year^{-1}$ for the Elgeyo and Nyambene non-use values, respectively. These estimates were high compared to a study in Endau Rompin National Park in Malaysia that estimated the NUVs at RM 31.46 (USD 6.91) $ha^{-1}year^{-1}$ (Nitanan et al., 2020). Equally, high compared to a study in East Mau which value NUVs at 1,017.60 (USD 9.51) $ha^{-1}year^{-1}$ (Langat, 2016) The study would attribute this to heterogeneous socio-economic and cultural particularly among forest-adjacent communities (Bamwesigye et al., 2020; Eregae et al., 2021). Also, attribute this to the difference in the level of understanding and perceived 'public good' nature of ES and valuation (Kakuru et al., 2013).

Table 4.36: Economic Valuation for Non-use Values (Bequest and Existence)

Ecosystem Service	Mean WTP (KES)	Elgeyo		Nyambene	
		No. of HH (Elgeyo)	Economic Value (KES)	No. of HH	Economic Value (KES)
Bequest	973.70	99,119	93,616,805.19	173,743	164,098,352.33
Existence	1,187.70	99,119	114,191,927.21	173,743	200,163,924.27
Total			207,808,732.40		364,262,276.59
Unit Value (KES)			8,237.92		67,121.12

4.2.4 Summary Total Economic Valuation

The study estimates the aggregated value for the two watersheds at KES 58.8 billion (USD 549.7 million) and KES 39.4 billion (USD 368.4 million) for the Elgeyo and Nyambene, respectively (Table 4.37). This translates to KES 542,793.97 (USD 5,072.84) and KES 1,3 million (USD 12,152.99) ha^{-1} $year^{-1}$. Albeit the aggregate value places the Elgeyo ecosystem higher, the Nyambene recorded a higher relative unit value compared to the former. The study attributes the higher aggregate of Elgeyo value to the size of the ecosystem. While it attributes the high unit value of Nyambene to, among others, a relatively 'well-conserved' native ecosystem and higher population density. The relative proportion placed indirect use values higher at 78% and 70% for the Elgeyo and Nyambene, respectively. Followed by direct use values at 22% and 30% of the total value, respectively. Globally, the findings were slightly lower for the case of Elgeyo but consistent for the case of Nyambene compared to a study in Florida forest valued at USD 12,424.1 ha^{-1} $year^{-1}$ (Escobedo & Timilsina, 2012).

The Elgeyo estimates were, however, slightly higher compared to estimates of a study in private forests in Georgia valued at USD 4,213.82 ha^{-1} $year^{-1}$ (Moore et al., 2011), and Texas forests valued at USD 3,677.83 (Simpson et al., 2013). Equally, lower for the case of Elgeyo and higher for Nyambene in comparison with estimates of a study in China, which valued TEV at USD 7,047 ha^{-1} $year^{-1}$ (Xi, 2009), and a study in Malaysia with TEV estimated at RM 266,864.30 (58,651.30) $ha^{-1}year^{-1}$(Nitanan et al., 2020). Locally, the relative unit area values were higher compared to TEV studies in Masai Mau, Transmara, and East Mau estimated at KES 116,120.21 (USD 1,085.24) $ha^{-1}year^{-1}$ (Kipkoech et al., 2011) and Mau complex, KES 363,336 (USD 3,398.48) $ha^{-1}year^{-1}$ Cherangany, KES 410,780 (USD 3,839.07) $ha^{-1}year^{-1}$ and Mt Elgon KES 1,066,790 (USD 9,970) $ha^{-1}year^{-1}$ (Langat et al., 2021; MoE & F, 2019a). The study would attribute the difference in unit values to a couple of factors, including inevitable changes in prices over time (Everard, 2009). Second, the difference in methodological approach and techniques, the scope, and forest ecosystem services incorporated in the study. For instance, most of the comparative studies didn't include the value of insect pollination, soil carbon sequestration, and non-use values, among others.

Table 4.37: Aggregated values for Ecosystem Services

Ecosystem Services	Elgeyo		Nyambene	
	Value (KES)	%	Value (KES)	%
Wood and Non-Wood Products	9,540,789,982.41	16.22	6,996,905,373.86	17.75
Domestic Water	1,001,467,893.15	1.70	1,299,211,315.61	3.30
Livestock Water	1,902,221,414.86	3.23	750,937,098.08	1.91
Cultural/Spiritual Values	125,686,132.65	0.21	2,388,999,961.33	6.06
Recreational Values	195,602,444.53	0.33	203,840,733.28	0.52
Direct Use Values	**12,765,767,867.60**	**21.70**	**11,639,894,482.16**	**29.53**
Watershed protection	15,644,985,024.48	26.60	8,671,035,541.16	22.00
Water purification	2,216,402,196.02	3.77	1,228,412,950.51	3.12
Soil Conservation	477,865,598.37	0.81	130,687,341.87	0.33
Nutrient Conservation	3,568,003,646.75	6.07	935,441,495.34	2.37
Plant CO_2 Sequestration	3,358,068,386.79	5.71	2,062,360,469.25	5.23
Soil CO_2 Sequestration	16,116,399,974.10	27.40	1,706,458,810.28	4.33
Oxygen generation	3,191,219,400.92	5.43	1,959,890,026.97	4.97
Microclimate influence	452,956,020.26	0.77	6,658,365,491.87	16.89
Pollination	821,477,008.61	1.40	4,061,292,428.18	10.30
Indirect Use Values	**45,847,377,256.31**	**77.94**	**27,413,944,555.43**	**69.55**
Bequest	93,616,805.19	0.16	164,098,352.33	0.42
Existence	114,191,927.21	0.19	200,163,924.27	0.51
Non-Use Values	**207,808,732.40**	**0.35**	**364,262,276.60**	**0.92**
Total Economic Values	**58,820,953,856.31**	**100.00**	**39,418,101,314.19**	**100.00**

4.3 Impact of land use/ land cover regime on stock and flow on selected ES

4.3.1 Trends in Land Cover (Elgeyo)

The Elgeyo land cover analysis exhibited varied trends and proportions. Based on Landsat imagery analysis, in 2019, wooded grassland was established as the dominant land cover at 46% of the total area. This was followed by cropland at 33% and forest at 12% of the total area. The time series land cover analysis exhibited significant changes over the last three decades. Where, for instance, cropland recorded a significant increase

($F_{(1,4)}$ =25.13, P=0.02) from about 8,000 hectares in 1990 to about 35,000 ha in 2019, translating to an approximately 350% increase. Dense forests declined significantly with $F_{(1,4)}$ =10.567, P=0.0474 from about 25, 000ha to 11,000 between 1990 and 2019, translating to a 56% decline, equivalent to approximately 2% of dense forest is lost to other land uses every year. Equally, the ecosystem recorded a decline in wooded and open grassland by approximately 7%, equivalent to a loss of 0.2% every year (Table 4.38). Further analysis revealed a significant decline in the overall tree cover with $F_{(1,4)}$ =27.36, P=0.014, from about 38,000ha in 1990 to about 25,000 ha in 2019 This translates to approximately 3,200ha of tree cover lost every ten years (Figue 4.3). A regression analysis on tree cover as a function of land cover change exhibited significant differences, $F_{(2,2)}$ =204.818, P=0.005. In that regard, a unit increase in a dense forest would increase tree cover by a factor of 0.48, while an increase in cropland in the ecosystem would reduce the tree cover by a factor of 0.24.

Table 4.38: Land Cover/ Land Use Change between 1990-2019 (Elgeyo)

Land Cover Category	1990	2000	2010	2019	(%) 2019
Dense Forest	24,855.30	18,648.00	22,262.85	10,764.10	9.95
Moderate Forest	193.50	720.18	556.56	2,588.81	2.39
Open Forest	4.32	10.53	25.47		-
Wooded Grassland	53,786.34	56,592.36	48,420.09	49,883.99	46.11
Bushland	21,176.37	11,958.48	13,347.54	7,083.08	6.55
Perennial Cropland	122.85	121.59	384.93	1,122.79	1.04
Annual Cropland	7,978.86	20,181.96	22,738.86	35,157.74	32.50
Vegetated Wetland	139.41	23.49	96.12	283.46	0.26
Open Water	20.97	34.29	97.02	52.50	0.05
Other lands	88.83	75.87	437.31	1,240.09	1.15
Total	**108,366.75**	**108,366.75**	**108,366.75**	**108,176.57**	100.00

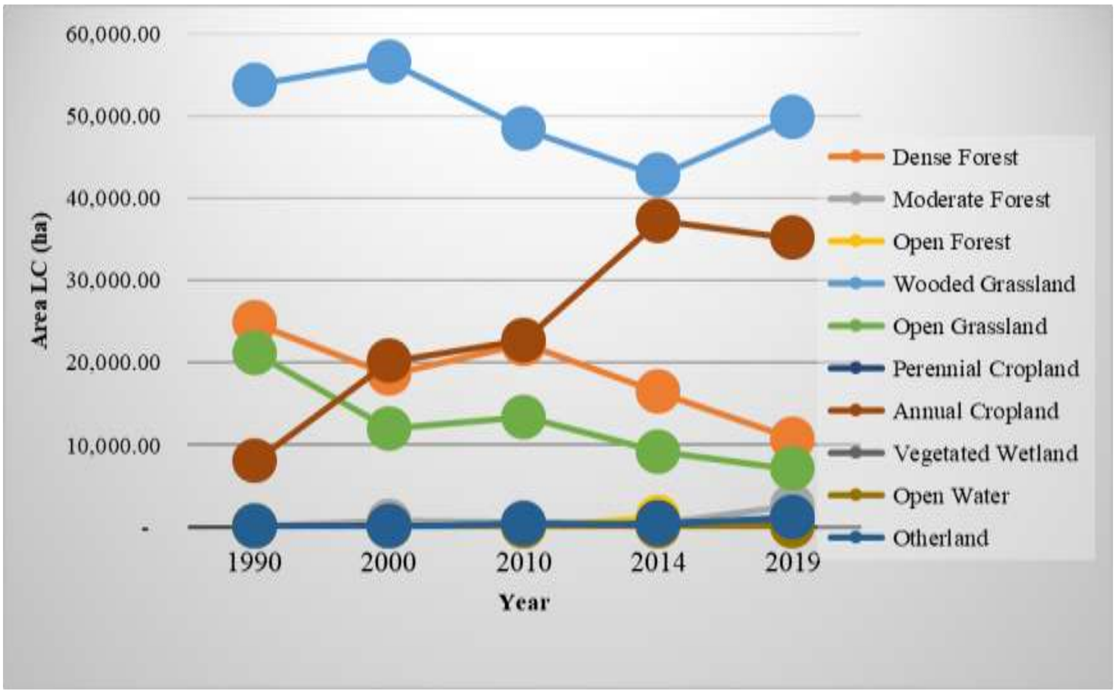

Figure 4.3: Elgeyo Land Cover Trends (1990-2019)

In terms of impact ranking, the study would rank cropland increase in terms of impact on tree cover (Table 4.39). The model explains approximately 99% of the variance in tree cover change over time as associated with the status of dense forest and cropland, as shown by the model Adjusted R^2=0.99. In that regard, the study forecast for tree cover change to between 19,876.98±2,432.81hectares to 13,334±2,656.71hectares for 2030 and 2070 respectively, other factors held constant. These findings agree with a study in Kibwezi at estimated a cropland increase of approximately 360% and a decline in forest land by 73% in ten years (Ruttoh et al., 2022). Equally, consistent with a study in the Mt Elgon water catchment with an increase of approximately 29% for cropland and a decline of 18% for natural forest (Masayi et al., 2021).

Table 4.39: Analysis of Factors Influencing the Change in Tree Cover

Model	R	R^2	Adj. R^2	Std. Error	Change Statistics				
					R2	F	df1	df2	Sig. F
1	0.998a	0.995	0.99	524.17	0.995	204.82	2	2	.005

a. Predictors: (Constant), Annual Cropland, Dense Forest

Model Coefficients [a]	Unstandardised Coefficients		Std Coefficients	t	Sig.	Correlations		
	B	Std. Error	Beta			Zero-order	Partial	Part
(Constant)	28651.9	2633.62		10.88	0.01			
Dense forest	0.479	0.091	0.49	5.26	0.03	0.96	0.966	0.26
Cropland	-0.243	0.042	-0.55	-5.85	0.03	-0.96	-0.97	-0.29

a. Dependent Variable: Tree Cover

4.3.1.1 Impact of the Land Cover Land Use Change on Forest Biomass (Elgeyo)

The forest carbon assessment for the Elgeyo ecosystem varied significantly ($F_{(7,47)}$ =3.597, P=0.004) across the different land cover. Dense forests recorded the highest average tree carbon, followed by moderately dense forests, with cropland land and glades recording zero values. The study estimated the overall mean at 56 mg of C/ha (Table 4.40). The assessment of the Elgeyo forest carbon over the last three decades recorded a significant difference, $F_{(1,3)}$ =40.795, P=0.008. It attributes the decline of forest carbon to a decrease in land cover over the last three decades. The study forecasts are that the Elgeyo ecosystem will lose about 77,000 mg of carbon, an equivalent of 277,200 Mg CO_2e released into the atmosphere every year. In that regard, between 2020 to 2060, the Elgeyo forest would record a decline in forest carbon from 7 million Mg to about 3.5 million Mg of carbon by season nine (Figure 4.4). This will translate to about 12.8 million Mg of CO_2e released into the atmosphere by 2060. The study findings were inconsistent with a study on the impact of land cover change on carbon trends in Kenya, which established an 8% decline in forest carbon between 2004 and 2016 (Nyamari & Cabral, 2021). Equally consistent with a study of tropical forests by Baccini et al. (2017) that recorded over 800 Tg loss of carbon every year attributed to forest degradation. However, they differed from a study in Taita Hills that recorded an increase in forest carbon, particularly within farmlands, that would ordinarily record a decline (Pellikka et al., 2018).

Table 4.40: The Forest Carbon Statistic Across Land Cover (Elgeyo)

Land Cover type	**Mean**	**N**	**Std. Deviation**
Dense forest	144.67	4	86.37
Moderate forest	80.14	7	37.11
Wooded grassland	82.16	13	81.29
Degraded forest	36.80	9	45.72
Bushland	25.15	6	24.26
Cropland	0.00	1	0.00
Glades	0.00	6	0.00
Other lands	0.43	2	0.00
Total	56.06	48	67.03
Species Diversity	2.67	39	2.39
Tree Cover (%)	32.01	39	27.59
Tree DBH	25.06	39	12.22
Tree Height (M)	16.98	39	4.94

ANOVA

			Sum of Squares	df	Mean Square	F	Sig.
Total Carbon (Mg/ha) * Land Cover type	Between Groups	(Combined)	81579.722	7	11654.246	3.597	.004
	Within Groups		129594.930	40	3239.873		
	Total		211174.652	47			

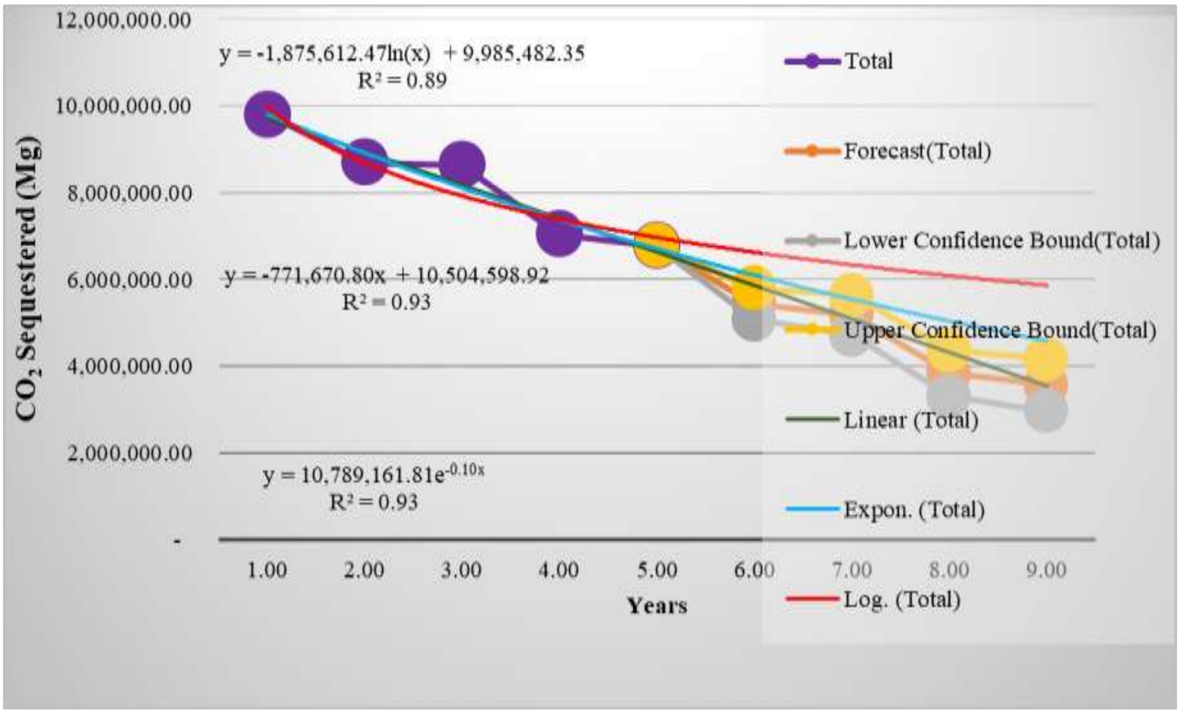

Figure 4.4: Forest Carbon Forecast (Elgeyo)

4.3.2 Trends in land cover and Land Use (Nyambene

Equally, the land cover in Nyambene exhibited varied proportions and trends in a ten-year basis analysis. In that regard, by 2019, cropland was recorded as the dominant land cover at 39%, followed by wooded grassland and dense forest at 25% and 24% of the total area, respectively (Table 4.41). However, the land cover change for Nyambene exhibited a non-significant difference on a ten-year basis analysis. Albeit an insignificant difference in land cover change decadally, there was a tree cover decline of about 214ha every ten years (Figure 4.5). Further scrutiny exhibits a significant decline in tree cover on a thirty-year basis analysis, whereby the intercept values recorded significance (t-test =5.74, $P<0.05$). This is a sign of a significant decline in the tree-covered area on a three-decade analysis basis with a significant impact on the stock and flow of ecosystem services in the future. Overall, the forest land cover changes were lower compared to most water catchment ecosystems in the country, which recorded significant changes. For example, a study in the Mau complex recorded a 15% decline in dense forestland and 12% in wooded grassland (Ayuyo & Sweta, 2014). Equally, changes were lower compared to a study in the Mt Elgon water catchment that established an 18%, 15%, and 16% decline in the natural forest, bamboo forest, and plantation respectively with 29% in cropland between 1999 and 2019 (Masayi et al., 2021).

Table 4.41: Nyambene Ecosystem Land Cover/ Land Use Change between 1990-2019

Sub-catchment	1990	2000	2010	2019	2019 (%)
Dense	8,328.14	7,358.23	6,382.32	7,157.03	23.61
Moderate	0.36	106.09	63.80	1,059.42	3.50
Open forest	-	208.93	0.36	-	-
Wooded grassland	5,258.49	5,868.41	1,166.39	7,662.31	25.28
Open grassland	288.55	779.07	159.05	3.60	0.01
Perennial cropland	8,489.27	6,569.13	4,005.43	1,902.64	6.28
Annual Cropland	8,943.20	10,387.42	19,538.43	11,761.26	38.81
Vegetated wetland	2.35	-	0.09	-	-
Open waters	0.36	7.32	0.63	-	-
Other lands	6.51	32.62	0.72	762.03	2.51
Total	31,317.23	31,317.23	31,317.23	30,308.30	100.00

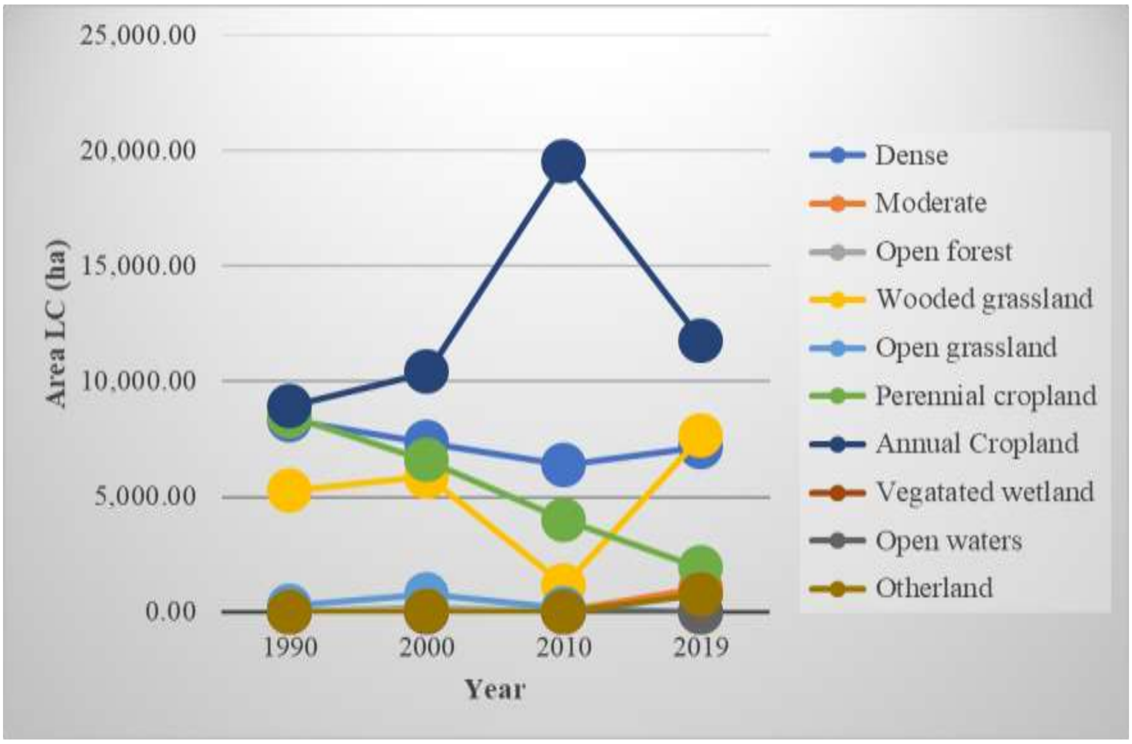

Figure 4.5: Nyambene land cover trends (1990-2019)

4.3.2.1 Impact of the Land Cover Land Use change on Forest Biomass (Nyambene)

The field-based forest carbon assessment recorded a significant difference across the land cover, $F_{(4,26)} = 7.182$, $p=0.0013$) in the Nyambene ecosystem. Dense forest recorded the highest mean tree carbon, followed by moderately dense forest and open forest with a mean of 210.47, 151.34, and 84.47 Mg of C/ha respectively, with an overall mean of 130 Mg of C/ha (Table 4.42). The analysis of both the Nyambene tree cover and forest carbon exhibited insignificant differences, $F_{(1,3)} = 0.07$, $P=0.8$ and $F_{(1,3)} = 0.34$, $P=0.6$ respectively. Albeit the insignificance on decadal, the findings exhibited significance on a forty-year basis with t-test= 8.2, $P<0.01$ and t-test=5.23, $P<0.01$ for forest carbon and tree cover, respectively. Overall, the findings show a slight decline, with an average loss of 121.7 ha of tree cover and about 65,752 Mg of carbon, an equivalent of 241,310 Mg of CO_2e released into the atmosphere every ten years. This translates to about 12ha of tree cover and about 6,600 Mg of forest carbon lost every year in the Nyambene ecosystem. Based on this trend, forest carbon would decline from 3 million to about 2,780,425.96±789,214.34 Mg between 2030 and 2060, respectively

(Figure 4.6). Although the decline of forest carbon is lower compared to the Elgeyo ecosystem, it is still consistent with studies such as the carbon trends in Kenya (Nyamari & Cabral, 2021). Likewise, similar to a study by Drigo (2018) that recorded a 2% annual decline in forest carbon in Kenya.

Table 4.42: The Forest Carbon Statistic Across Land Cover (Nyambene)

Land Cover Description	Total Carbon (Mg/ha)		
	Mean	**N**	**Std. Deviation**
Dense Forest	210.465	10	100.4067
Moderate Forest	151.341	8	67.9548
Open Forest	84.472	1	
Open Areas	20.293	6	18.7583
Savana Woodland	24.071	2	32.1636
Total Carbon (Mg/ha)	130.023	27	103.6800
Species Diversity	9.556	27	5.2134
Tree Cover%	52.751	27	34.9925
DBH	64.124	27	31.5625
Tree Height	13.996	27	3.8233
Wood Density (g/cm3)	.474	27	.1315

ANOVA			Sum of Squares	Df	Mean Square	F	Sig.
Total Carbon (Mg/ha) * Land Cover Description	Between Groups	(Combined)	158276.34	4	39569.09	7.18	.001
	Within Groups		121211.79	22	5509.63		
	Total		279488.13	26			

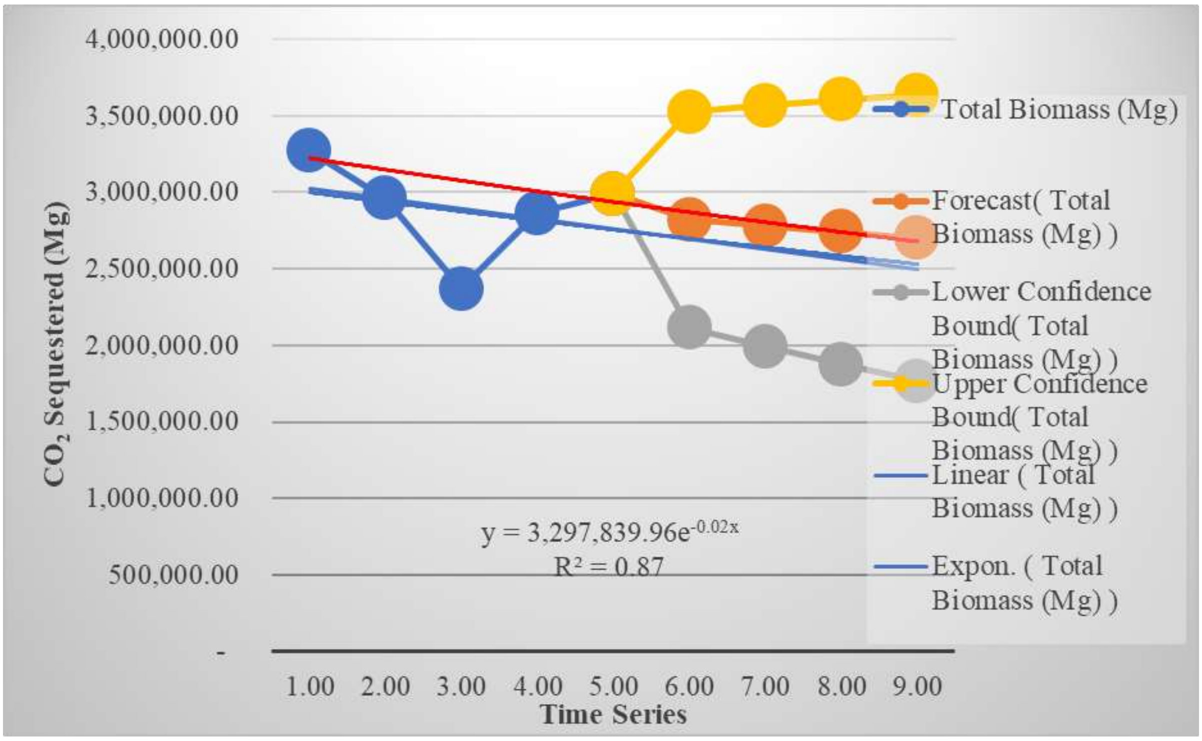

Figure 4.6: Forest Carbon Forecast (Nyambene)

4.3.3 Impact of Land Cover Change on River Flow Dynamics

The study assessed the water level and discharge for one major basin in Elgeyo (Moiben- station 1BA01) and two in Nyambene (Ura- station 4F09 and Thananthu- station 4F20) between 1953 and 2018. The assessment of the river flow based on WRA flow data shows that Moiben basin monitoring station 1BA01 recorded a minimum water level (0.2m), a maximum (1.45m), and a mean (std deviation) of 0.36 (0.13) m). Equally, it records a minimum discharge of 0.16 m^3/s, a maximum discharge of 34.28 m^3/s, and a mean (std deviation) discharge of 1.07 (1.90) m^3/s. While in the Ura basin station (4F09), records a minimum water level of 0.01m, a maximum of 2.86m, and a mean (std deviation) of 0.59 (0.44) m. Likewise, records a minimum discharge of 0.01, a maximum of 68.82 m^3/s, and a mean (std deviation) discharge of 6.3 (9.77) m^3/s. Thananthu basin station (4F20) records a minimum water level of 0.01, a maximum of 3.12m, and a mean (std deviation) of 0.45 (0.25) m. While it records a minimum discharge of 0.001 m^3/s 1, a maximum of 50.95 m^3/s, and a mean (std deviation) of 4.24

(4.91) m^3/s (Table 4.43). The river flow analysis exhibited data annual variation for both the water level and discharge. However, stations in the Nyambene ecosystem recorded higher river flow dimensions compared to the Elgeyo basin. The changes in average river flow attributed to tree cover exhibited a significant difference with $F_{(1,46)} = 6.629$, $P<0.05$, $F_{(1,47)} = 7.629$, $P<0.05$, $F_{(1,47)} = 10.270$, $P<0.05$ for Moiben, Ura and Thananthu rivers respectively. The study estimates decline associated with tree cover change at 0.004mm, 0.07mm, and 0.07mm annually, which translates to 0.3%, 1.1%, and 3% of the average flow for Moiben, Ura, and Tananthu, respectively. The proportion of change is lower compared to a study in Burkina Faso that estimated the impact of land on rivers at between a 12% and 95% increase in peak flows and a decline of between 24% and 44% in dry season flow (Idrissou et al., 2022).

Table 4.43: River Flow Statistics

Basin Stations	Dimensions	N	Min	Max	Mean	Std. Dev
Moiben1BA01	Water Level	6456	0.20	1.45	0.36	0.13
	Discharge	6456	0.16	34.28	1.07	1.90
Ura 4F09	Water Level	6456	0.01	2.86	0.59	0.44
	Discharge	6456	0.00	68.82	6.30	9.77
Thananthu 4F20	Water Level	6456	0.01	3.12	0.45	0.25
	Discharge	6456	0.01	50.95	4.24	4.91

4.3.3.1 River Water Level

The analysis of the average water level change over the years exhibited a significant difference for both stations. It exhibits this by $F_{(1,63)} = 17.614$, $P<0.01$; $F_{(1,60)} = 35.534$, $P<0.05$; and $F_{(1,51)} = 5.429$, $P<0.05$ for Moiben, Ura and Thananthu basins, respectively. Adjusted R^2 values of 0.21, 0.36, and 0.08 can explain the variance in the average water level change over time by 21%, 36%, and 8%, respectively. The study estimates the decline of the average water level at 1.4mm, 6.5mm, and 2.4mm annually for Moiben, Ura, and Thananthu, respectively (Figure 4.7). The assessment of the influence of land cover change (tree cover) on the river water level in that regard exhibited a significant difference with $F_{(1,46)} = 6.629$, $P<0.05$, $F_{(1,47)} = 7.629$, $P<0.05$, $F_{(1,47)} = 10.270$, $P=0.002$ for Moiben, Ura, and Thananthu rivers respectively. A reduction in water levels associated with the alteration of tree cover was calculated to be 0.004mm, 0.07mm, and 0.07mm per year, respectively. The study explains the attributed tree cover change by 11%, 12%, and 11% as exhibited by the model adjusted R^2= 0.107, 0.121, and 0.112 respectively. According to the findings, every decrease in one-hectare tree cover leads to a decline in average water levels. Although not comparable in terms of numbers, declining water levels over the years are consistent with a study of the Weruweru-Kiladeda sub-catchment in the Pangani River Basin, Tanzania that recorded low dry season flows and peak wet seasons (Chiwa, 2012). Equally, they agree with a study in East Mau, which also suggested that deforestation and land use change result in reduced stream flows and subsequent water shortages (Kundu et al., 2004) and a study in Upper Mara River (Mwangi et al., 2016).

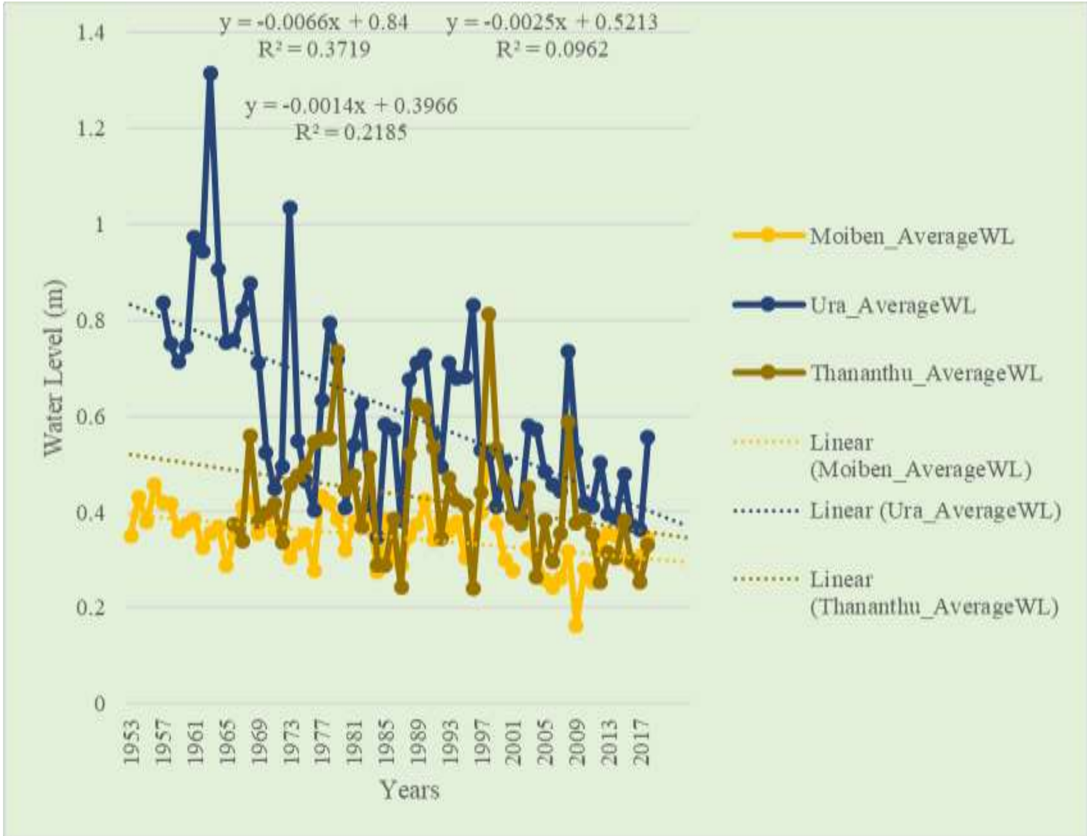

Figure 4.7: The Average Water Level (m) Trends (1953 to 2018)

Equally, the study assessed the peak water level, whereby the regression analysis on the peak water level over time and, as a function of tree cover change, exhibited significance for the monitoring except for the Moiben basin. The analysis of peak water levels over the years records a significant increase over time (Figure 4.8Figure). In the analysis, there was a significant increase in Ura and Thananthu rivers, with $F_{(1,47)} =5.566$, $P<0.05$ and $F_{(1,47)} =11.580$, $P<0.05$, respectively. The increase in peak flow attributed to tree cover change is estimated to be 0.3mm and 0.4mm for the Ura and Thananthu rivers, respectively. This suggests that every decline in tree cover will lead to an increase in peak river water level by 0.3mm and 0.4mm respectively. These as explained by 9% and 18% for Ura and Thananthu, as exhibited by adjusted $R^2= 0.087$ and 0.181, respectively. The findings suggest that deforestation, particularly in catchment areas, would lead to reduced base flows consistent with the findings reported by a couple of studies (Chang, 2007; Dougherty et al., 2007). Equally in agreement with a study in the Malewa River watershed (Cheruiyot et al., 2020), and a study in the Nyando River sub-catchments (Kundu & Olang, 2011).

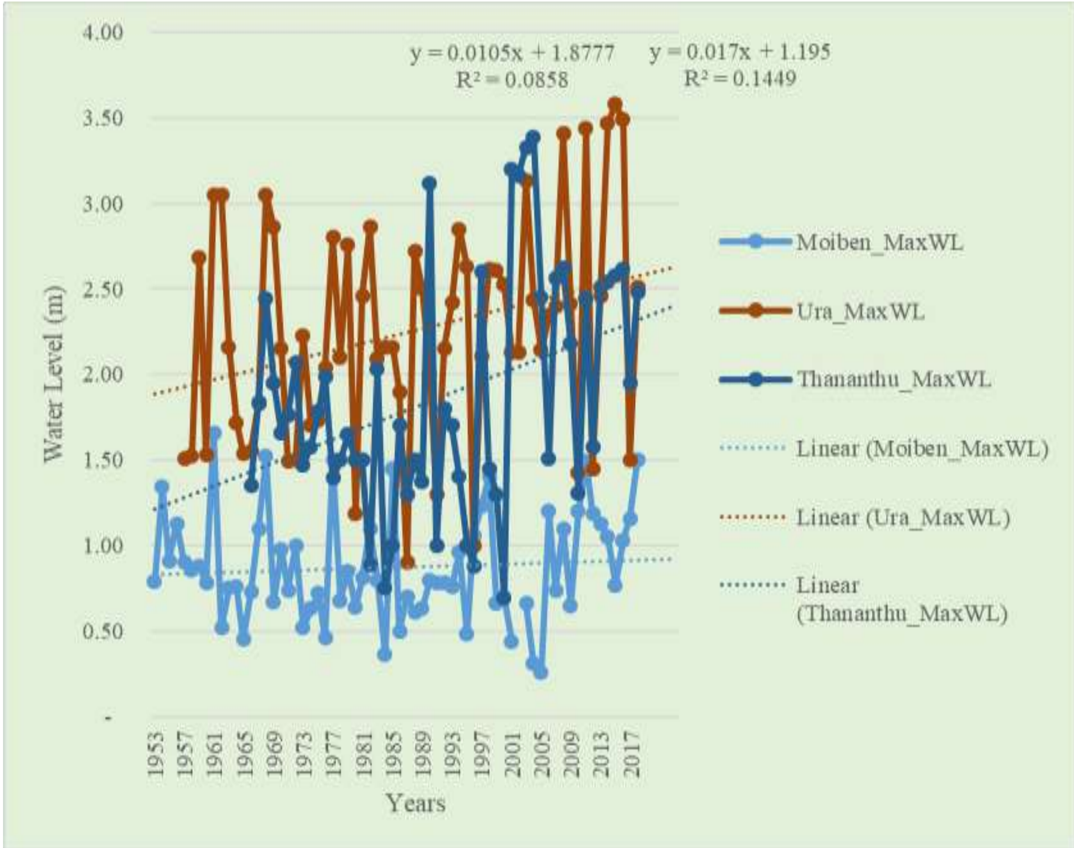

Figure 4.8: The Peak River Water Levels (m) Trends (1953 to 2018)

4.3.3.2 River Discharge

The analysis of base flow volume over time recorded varied significance except for the Thananthu River. A significant decrease in the base flow of Moiben and Ura rivers over time was observed with $F_{(1,64)} = 5.686$, $P=0.020$ and $F_{(1,60)} = 4.727$, $P=0.034$, respectively. The study estimates the decline at $1mm^3$/sec and $10mm^3$/sec for Moiben and Ura, respectively. The model can account for 6.7% and 5.8% of the variance in the base flow discharge due to time change, as shown by the adjusted R^2 values of 0.067

and 0.058, respectively (Figure 4.9). Analysis of base-flow discharge change was significantly affected by the tree cover change test, as evidenced by $F_{(1,47)}$ =17.934, P<0.05 and $F_{(1,47)}$ =24.314, P<0.05 for Moiben and Ura, respectively. Tree cover change was found to cause a decrease in the lowest recorded discharge by $0.008mm^3/sec$ and $0.34mm^3/sec$, respectively, according to the study's estimate. Base flow decline attributed to tree cover decrease explained by the model is about 26.1% and 32.7% for Moiben and Ura respectively, as stated in the model adjusted R^2 0.261 and 0.327 respectively.

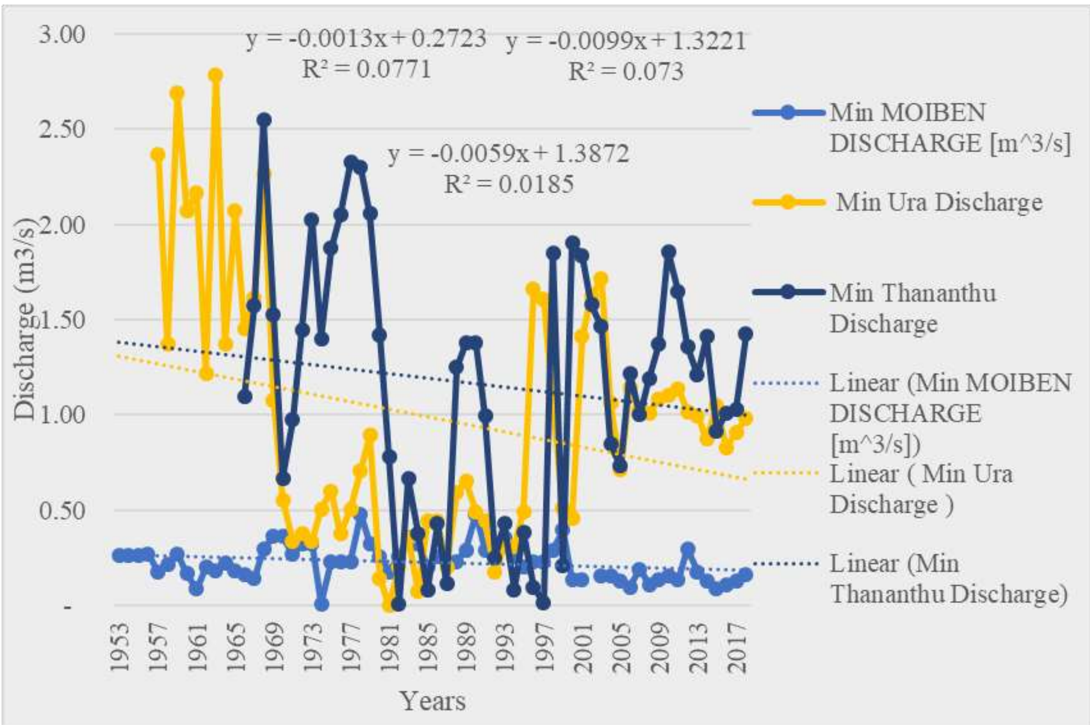

Figure 4.9: Simulated Base Flow (Low-Discharge) (m^3/s) Between 1953 to 2018

Although the analysis on peak discharge recorded insignificant differences, the average river discharge as a function of time series generated varied outputs in the three river basins assessed (Figure 4.10). The regression analysis exhibited a significant increase in the average water volume for two rivers (Moiben and Thananthu) while a significant decline in average water volume in the Ura river, with $F_{(1,64)}$ =11.311, P=0.001 (Moiben), $F_{(1,51)}$ =12.971, P=0.001 (Thananthu) and $F_{(1,60)}$ =12.746, P=0.001 (Ura). The study can estimate the increase in average river discharge over time at 16mm^3/sec (Moiben) and 52mm^3/sec (Thananthu) and a decline of 69mm^3/sec (Ura). According to the study, the variance attributed to time change was 13.7%, 16%, and 18.7% for Moiben, Ura, and Thananthu. It exhibits this by the model adjusted R^2 equivalent to 0.137, 0.161, and 0.187. The study could attribute the decline in the average discharge for the Ura River to a couple of factors, including water abstraction from the four dams constructed along the river system that diverts most of the water. A test on the change in the average river discharge as a function of tree cover change exhibited significance for Moiben and Thananthu with $F_{(1,47)}$ =6.084, P=0.017, and $F_{(1,47)}$ =5.109, P=0.028, respectively.

However, there was no significant difference recorded in the average discharge attributed to tree cover change for Ura. This study found that the decrease in tree cover caused one river to decline by 0.05mm^3/sec, while the other increased by 1mm^3/sec. The research calculates the variation in the mean river discharge over time linked to modifications in tree cover at 10% and 8%, which resulted in a model-adjusted R^2 of 0.096 and 0.079 in Moiben and Thananthu, respectively. Any decrease in tree cover per unit area would result in a significant increase in the average river discharge, as specified in the findings of Moiben and Thananthu. Overall, the findings suggest that the conversion of forests to other land uses, particularly in catchment areas, would lead to enhanced surface run-off, enhanced peak flows, and reduced base flows. This was consistent with a study undertaken by Chebet et al. (2017) in the Elgeyo catchment that showed river discharge increases with increased deforestation. Equally, consistent with a study in Portland Metropolitan Area, Oregon, USA (Chang, 2007) and another in Washington's western watersheds (Dougherty et al., 2007). Likewise, in agreement with a study in the Upper Mara River (Mwangi et al., 2016), and the Ragati River sub-catchment (Mwangi et al., 2019). Both studies established that the degradation of a water catchment ecosystem will result in increased peak and flood volumes and reduced base flows.

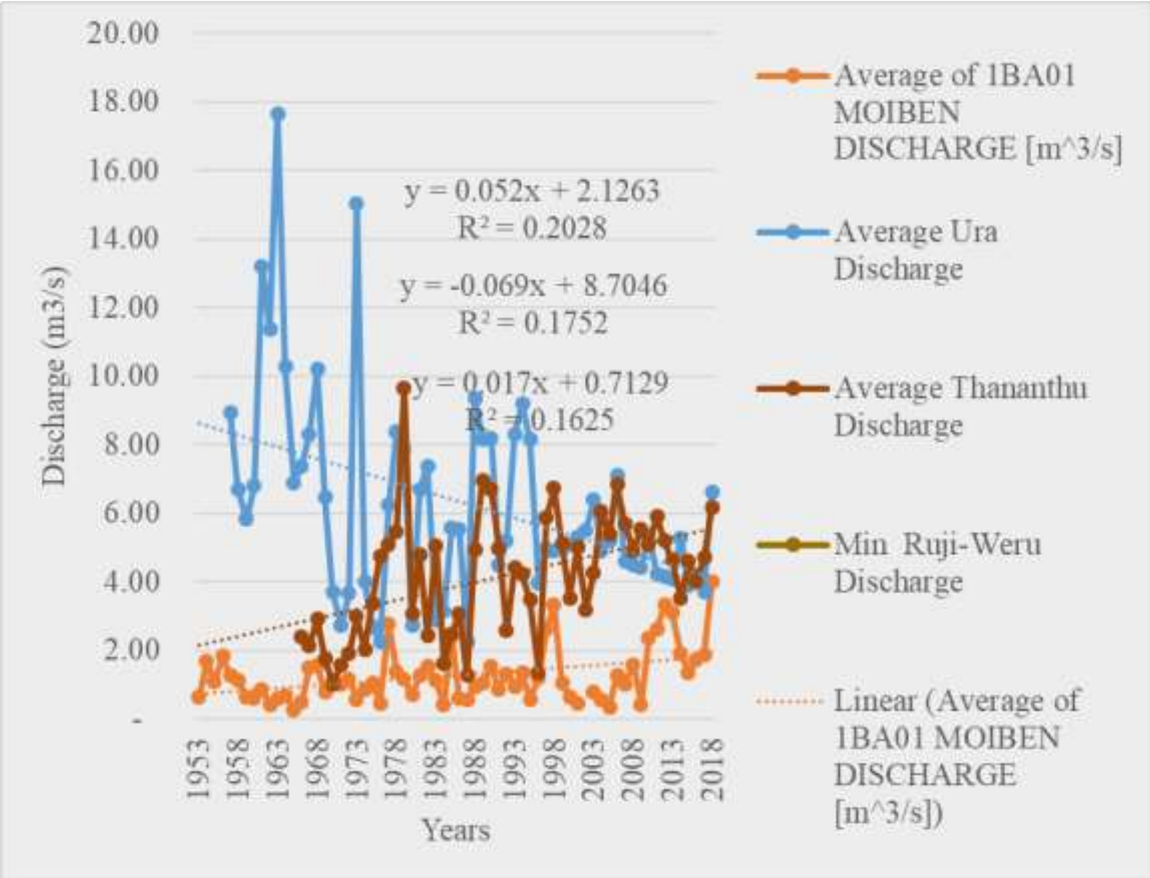

Figure 4.10: Simulated Average River Discharge Between 1953 to 2018

4.4 Modelling the impact of land use/cover change on stock and flow of ES

4.4.1 Modelling Forest Biomass (Elgeyo)

The study employed the traditional Gamma GLM log link to the model, an area unit-based forest biomass for the Elgeyo ecosystem. It placed Chave et al. (2014) algorithms generated estimates as a function of land cover, species diversity, and tree volume. The results revealed that land cover, species diversity, and stem volume significantly influence the mean unit area biomass for the Elgeyo ecosystem at a factor of 2.9, 1.3, and 1.5, respectively (Table 4.44). This output suggests that an increase in a unit of forest cover increases the mean forest carbon by a unit of 2.9, while a unit increase in species diversity and stem volume would increase forest carbon by a unit of 1.3 and 1.5, respectively. Based on the standard coefficients, the study would rank land cover followed by species diversity and stem volume in that order in terms of the effect degree on forest biomass. Using the Gamma GLM (Log link) output, the study generated a unit-based algorithm model, as shown (4.1).

Table4.44: Elgeyo Biomass Gamma GLM (Log link)

Tree carbon (Mg/ha)	Coef.	Std Err.	t-value	p-value	[95% Conf. Interval]		Sig
Land cover	2.886	.376	8.14	0	2.236	3.724	***
Species diversity	1.259	.081	3.59	0	1.11	1.427	***
Stem volume (m3)	1.502	.116	5.26	0	1.291	1.748	***
Constant	.491	.223	-1.57	.117	.201	1.195	
Mean dependent var		60.441	SD dependent var				61.686
Number of obs		39	Chi-square				99.490
Prob > chi2		0.000	Akaike crit. (AIC)				360.921
*** *p<.01*, ** *p<.05*, * *p<.1*							

$$\log_{10}Y=(0.49\pm0.22)\log_{10}(1)+(2.89\ (\pm0.38)\log_{10}(LC)+(1.26\ (\pm0.08))\log_{10}(species)+(1.5\ (\pm0.12))\log_{10}(stem\ vol) \quad 4.1$$

4.4.1.1 Model Test (Elgeyo)

The study tested model-generated estimates by comparing them with Chave et al. (2014) generated estimates. The output exhibited a non-significant difference with $F_{(2,110)} = 3.07$ $p>0.05$ (Table 4.45). This validates similarities in the models, however, it exhibited an overestimation of the biomass for sample points with variables with the highest scores. For instance, a land cover score >3, stem volumes > $3m^3$, and species diversity >5 recorded extremely high estimates. This suggests the model works well in a landscape with a moderate score of explanatory variables and not very well in a diverse and dense forest. Thus, though the researchers can deploy a model in a similar watershed, they should consider the limitation cited. Application of the model should, however, code land cover on a scale of 0 to 5. Glades and cropland (0), other lands (1), bushland (2), woodland (3), moderate (4), and dense forest (5). Based on the analysis output, the model differs by approximately 38% compared with Chave et al. (2014) model. The difference in the two models is within the range of a comparison between the random forest and stratified approach model, which recorded between 43% and 37% in comparison with airborne LiDAR validation area estimates (Mascaro et al., 2014). However, the difference between the two models is higher compared to a difference of 10% recorded in a study comparing the European Forest Information Scenario Model (EFISCEN 4.2.0) and the Carbon Budget Model of the Canadian Forest Sector (CBM) (Blujdea et al., 2021).

Table 4.45: Model 1 Test (Elgeyo) Descriptive Statistics

Variable	Obs	Mean	Std. Dev.	Min	Max
Chavel et al.(2014) (Mg/ha)	39	60.441	61.686	0	251.461
Prediction (Mg/ha)	39	76.994	99.2	0	423.71
Model Test (Mg/ha)	35	83.354	113.383	0	548.744
Single-factor ANOVA, $F_{(2,110)}$ = 3.07, P=0.55					

4.4.2 Modelling Forest Biomass (Nyambene)

The study employed the traditional Gaussian GLM log link to the model, an area unit-based forest biomass for the Nyambene ecosystem. It placed the Chave et al. (2014) generated biomass as a function of land cover, Number of stems, species diversity, and tree volume. The analysis reveals that land cover, species diversity, and stem volume significantly influence the mean unit area biomass at a score of 1.35, 1.05, and 1.01, respectively (Table 4.46). This suggests that an increase in a unit of forest cover increases the mean forest carbon by a unit of 1.3, while a unit increase of species and stem volume would increase forest carbon by a unit of 1.05 and 1.01, respectively. Based on the standard coefficients, the study would rank land cover the highest in terms of the effect degree on forest biomass. Based on the gamma GLM log link output, the study developed an area unit-based biomass predictive model as shown (4.2).

Table 4.46: Nyambene Biomass Gaussian GLM (log link)

Tree Carbon (Mg/ha)	Coef.	St. Err.	t-value	p-value	[95% Interval]	Conf.	Sig
Land cover	1.345	.139	2.87	.004	1.098	1.648	***
Species diversity	1.046	.018	2.57	.01	1.011	1.083	**
Stem volume (m3)	1.014	.003	4.28	0	1.007	1.02	***
Constant	18.116	7.838	6.69	0	7.758	42.301	***
Mean dependent var		130.023	SD dependent var				103.680
Number of obs		27	Chi-square				52.114
Prob > chi2		0.000	Akaike crit. (AIC)				295.971
*** *p<.01*, ** *p<.05*, * *p<.1*							

$$\log_{10}(Y)=18.1(\pm 7.84)\log_{10}(1)+\left(1.35(\pm 0.14)\right)\log_{10}(LC)+\left(1.05(\pm 0.02)\right)\log_{10}(\text{species})+\left(1.01(\pm 0.003)\right)\log_{10}(\text{stem vol}) \quad 4.2$$

4.4.2.1 Model Test (Nyambene)

The t-test between the model-predicted estimates and Chave et al. (2014) generated estimates exhibited a non-significant difference with $F_{(2.69)} = 3.12$ P=0.94 (Table 4.47). Worth noting the model test exhibited extremely higher values on explanatory variables with higher scores. For instance, a sample point with a land cover score >5, stem volumes > 5m, and species diversity >11 recorded higher forest biomass estimates. This suggests the model works well in a landscape with a moderate score of explanatory variables and with minimal widespread data and extreme scores. Thus, any deployment of the model should consider the ecosystems with a moderate score of the quoted explanatory variables, data spread, and standard error. Based on the analysis out, the model estimates differ by 4.6% in terms of the average unit area biomass compared with the generalised improved allometric model by Chavel et al. (2014). The difference is lower compared with a study comparing the European Forest Information Scenario Model (EFISCEN 4.2.0) and the Carbon Budget Model of the Canadian Forest Sector (CBM) (Blujdea et al., 2021). Equally, lower compared to the difference recorded Random Forest and Stratification approach (Mascaro et al., 2014).

Overall, the study didn't undertake the sensitivity analysis since it was beyond its objective and which confirms the potentiality of the model overestimating biomass in a more diverse ecosystem. Although the study would recommend the application of a model in determining area unit-based forest biomass locally, it underscores the necessity of enhanced input-data quality and further data collection aimed at reducing the error margin and improving the model. However, the preference and adoption of the model would be primarily on suitability and the precision of the required data (Blujdea et al., 2021; Tedeschi, 2004).

Table 4.47: Model Test Descriptive Statistics (Nyambene)

Variable	**Obs**	**Mean**	**Std. Dev.**	**Min**	**Max**
Chave et al. (2014) (Mg/ha)	27	130.023	103.68	1.327	393.909
Predicted Carbon (Mg/ha)	27	132.373	86.623	31.16	355.804
Model Test Carbon (Mg/ha)	21	136.001	107.378	.32	356.885
Single-factor ANOVA $F_{(2.69)}$ = 3.12, P=0.94.					

4.4.3 Modelling river discharge

4.4.3.1 Moiben Sub-basin (1BA01)

The study employed VAR (1) on base flow, average flow, and peak flow and tree cover for Moiben (1BA01) and tree cover change. The Lagrange multiplier test exhibited non-significance, suggesting that the model records nil autocorrelation. While a test on the stability using Eigenvalue satisfies the condition with all the modulus values generated, recording a score less than one, suggesting a stable model. The Granger causality Walt test exhibited varied output on the influence of the four variables on each other. Overall, the Elgeyo the tree cover change does Granger causality the influence base flow as exhibited with $X^2_{(1)} = 5.682$ $p<0.05$ on Moiben sub-basin. However, tree cover change does not Granger causality influence peak flows at 95% confidence exhibited by $X^2_{(1)} = 2.654$, $p>0.05$ in sub-basin (Table 4.48).

Table 4.48: VAR (1) Model diagnostic (Moiben-1BA01)

Vector autoregression (1)			No. of obs	= 63
Sample: 1956 - 2018			AIC	= 10.49527
			HQIC	= 10.76286
Log-likelihood = -310.601			SBIC	= 11.17563
FPE = 0.425327				
Det (Sigma_ml) = 0.22511				
Equation	Parms	RMSE	R-sq	Chi2
Aver. discharge	5	.737569	0.2073	16.47617
Min discharge	5	.093778	0.4115	44.04932
Max discharge	5	9.65909	0.1185	8.472711
Tree cover	5	1.98586	0.1523	11.31543
Lagrange-multiplier test				
Lag	chi2		Df	Prob>Chi2
1	23.366		16	0.104
2	13.449		16	0.640
H0: no autocorrelation at lag order				
Eigenvalue stability condition				
All the eigenvalues lie inside the unit circle. VAR satisfies stability conditions.	Eigenvalue		Modulus	
	.4244306 +.2543469i		.494807	
	.4244306 -.2543469i		.494807	
	-.4285077		.428508	
	.0554204		.05542	
Granger causality Wald tests				
Equation	**Excluded**	**chi2**	**Df**	**Prob>Chi2**
Aver. Discharge	Min Discharge	2.954	1	0.086
	Max Discharge	1.226	1	0.268
	Tree Cover	0.026	1	0.873
	ALL	3.135	3	0.371
Min Discharge	Average Discharge	7.149	1	0.007
	Max Discharge	0.181	1	0.670
	Tree Cover	5.682	1	0.017
	ALL	33.087	3	0.000
Max Discharge	Average Discharge	5.149	1	0.023
	Min Discharge	2.769	1	0.096
	Tree Cover	2.654	1	0.103
	ALL	7.363	3	0.061
Tree Cover	Average Discharge	4.790	1	0.029
	Min Discharge	1.153	1	0.283
	Max Discharge	3.310	1	0.069
	ALL	10.695	3	0.013

The study explains approximately 15% of estimates regarding the level of influence of the various parameters in the model attributed to tree cover change as exhibited by adjusted R^2 (0.15) and RMSE (2.0). Likewise, the model constant was significant in all the variables, thus relevant in model development. Based on the VAR(1) lag values, the study generated a linear model for the respective variables as shown in equation 4.3, 4.4, 4.5.and 4.6.

$$\text{Av dis}_{1_t}=0.96_1+[(0.771_{11})(t\text{-}1_{\text{av dis}})]+[(-1.51)_{12})(t\text{-}1_{\text{min dis}})]+[(-0.023)_{13}(t\text{-}1_{\text{max dis}})] \quad 4.3$$

$$+[(0.007_{13})(t\text{-}1_{\text{tree cover}})]+0.28_{1_t}$$

$$\text{Min dis}_{1_t}=0.10_1+[(0.10)_{11}(t\text{-}1_{\text{av dis}})]+[(1.14)_{12}(t\text{-}1_{\text{min dis}})]+[(-0.001)_{13}(t\text{-}1_{\text{max dis}})] \quad 4.4$$

$$+[(-0.014_{13})(t\text{-}1_{\text{tree}})]+0.04_{1_t}$$

$$\text{Max dis}_{1_t}=10.82_1+[(8.31_{11})(t\text{-}1_{\text{av dis}})]+[(-19.15_{12})(t\text{-}1_{\text{min dis}})]+[(-0.43_{13})(t\text{-}1_{\text{max dis}})] \quad 4.5$$

$$+[(0.98_{13})(t\text{-}1_{\text{tree cover}})]+3.66_{1_t}$$

$$\text{Tree Cover}_{1_t}=0.46_1+[(-1.65_{11})(t\text{-}1_{\text{av dis}})]+[(-2.54_{12})(t\text{-}1_{\text{min dis}})]+[(0.103_{13})(t\text{-}1_{\text{max dis}})] \quad 4.6$$

$$+[(-0.006_{13})(t\text{-}1_{\text{tree cover}})]+0.75_{1_t}$$

Av dis (average discharge/average flow), Min dis (minimum discharge/base flow), and Max dis(maximum discharge/peak flow)

4.4.3.2 Model Test (Moiben)

The study ran a t-test on the significance between the model predicted with the actual river flow, whereby all the variables exhibited a non-significance difference with unpaired t-test p-value> 0.05 (Table 4.49). This suggests that the model-predicted values are not statistically different from the actual values. Equally, the residual mean recorded a zero score (Figure 4.11) which suggests that the errors for the predicted values are 'white noise' and thus in compliance with the VAR model assumption. The test on the tree covers impulse response (IRF) as a function of base flow, average flow, and peak flow exhibited varied output.

Table 4.49: Summary of descriptive statistics on river flow data (actual and predicted)

Variable	Obs	Mean	Std. Dev.	Min	Max	T-test, unpaired
Average discharge	65	1.264	.791	.286	3.981	t=0.385, df=126,
Pred Av discharge	63	1.26	.365	.515	2.365	p=0.97
Min discharge	65	.263	.116	.011	.678	t=0.007 df=126,
Pred Min discharge	63	.263	.076	.161	.533	p=0.9945
Max discharge	65	10.712	10.156	.361	37.5	T=0.244 df=126,
Pred Max discharge	63	10.383	3.426	3.314	21.386	p=0.808
Errors	63	0	.713	-1.309	2.307	

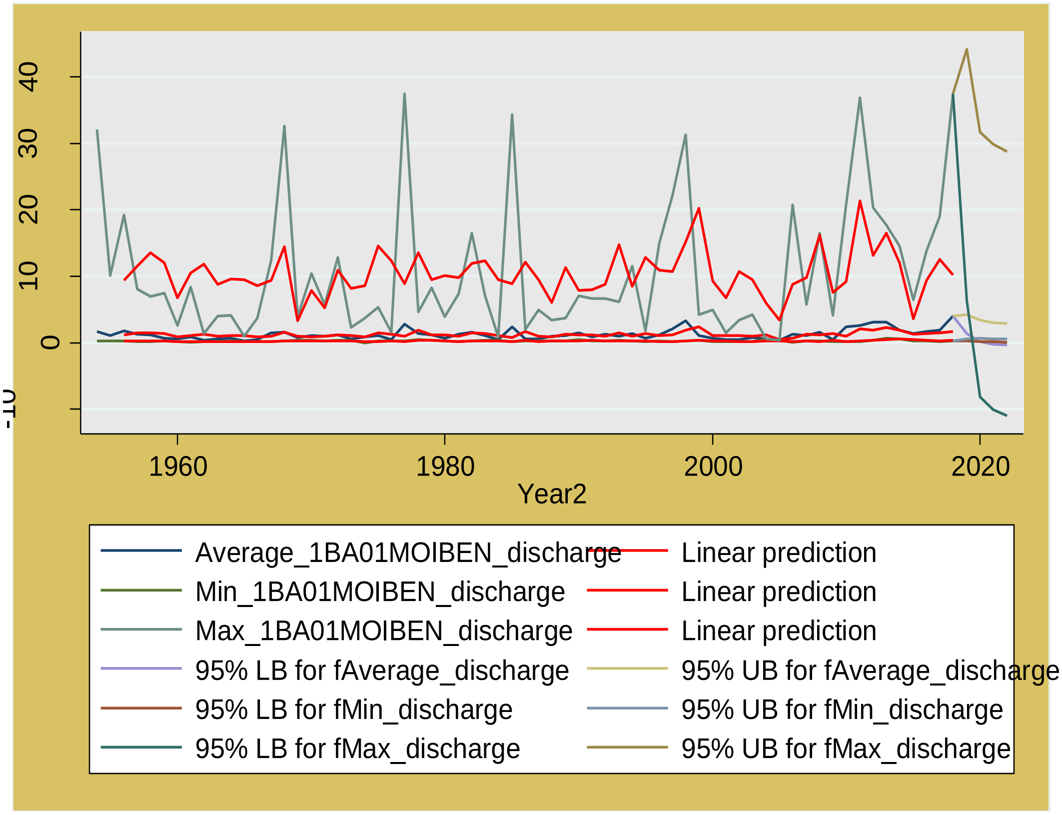

Figure 4.11: Time series line plot for Moiben (1BA01) river flow data

For instance, a unit increase in tree cover would surge the peak flow for the first two years, then decline for the next two years before it steadily declines (Figure 4.12). The tree covers impulse based on forecasted error variance decomposition (FEVD) recorded up to 5% and 4% for base and peak flow respectively (Appendix IX). This is a suggestion that impulse change on the Elgeyo tree cover can influence both base and peak flow to between 4 and 5% at a 95% confidence level.

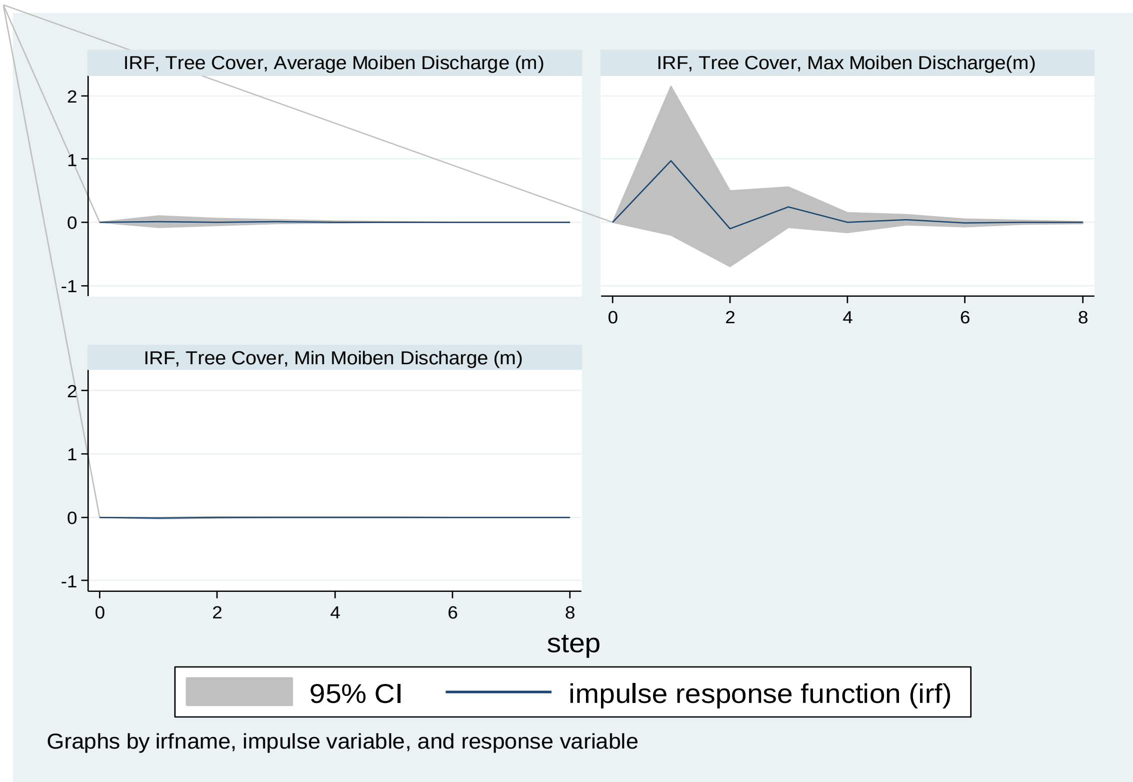

Figure 4.12: Moiben (1BA01) IRF, Impulse (tree cover), Response (river flow)

The findings were slightly lower compared to a study by Oztürk et al. (2013) that modelled land cover and use change on hydrology using the MIKE-SHE computer model (DHI, 2005). Equally, lower compared with a study in Thiba that recorded adjusted R^2 values of 0.7 and 0.9 using the Soil and Water Assessment Tool (SWAT) (Kasuni & Kitheka, 2017). Despite inconsistent model statistical output, the study findings on the impact of tree cover change attributed to land cover on peak flow are contrary to most of the literature, which commonly report that increase in forest cover result in an increase base flow and decline on peak wet season flow (Githui et al., 2009; Kasuni & Kitheka, 2017; Leta et al., 2018).

Based on the model, the tree cover change is not a primary predictor of peak flows, albeit being a good predictor of base flow. Equally, the findings suggest other bio-geophysical attributed such as erosivity, evapotranspiration, topography and human related drivers that influence river flow in the Moiben sub-basin, Elgeyo ecosystem. The latter includes damning, over-abstraction, and irrigation, among others, as reported by a couple of studies conducted in such similar water basins (Gao et al., 2013; Milliman et al., 2008).

4.4.3.3 Modelling of URA Sub-Basin (4F09)

Using the VAR (3) model for URA (4F09), the lagged values exhibited varied responses. Similarly, the Lagrange multiplier test exhibited non-significance, rejecting the null hypothesis on autocorrelation between the model lags. The Johansen test with four ranks satisfies the condition of no cointegration between lags. This is because a majority of trace statistics recorded higher values than critical values at 95% CL. The test on Eigenvalue generated modulus values less than one a suggestion that all the Eigenvalues line inside the unit circle and the VAR (3) model satisfies the stability condition. The Granger causality Walt test exhibited varied output on the influence of the four variables on each other. Overall, the tree cover does not Granger Causality cause both base flow and peak flow at a 95% confidence level, with $X^2_{(3)} = 0.753$, $P>0.05$ and $X^2_{(3)} = 6.794$, $p=0.079$, respectively (Table 4.50).

Table 4.50: Ura (4F09) VAR (3) Model Statistics

Vector autoregression lag (3)			No. of obs. = 36		
Sample: 1961 - 1996			AIC = 17.09861		
			HQIC = 17.89695		
Log-likelihood = -255.7751			SBIC = 19.38592		
FPE = 359.0717					
Det (Sigma_ml) = 17.43043					
Equation	Parms	RMSE	R-sq	Chi2	p>chi2
Aver discharge	13	3.18243	0.6378	63.40037	0.0000
Min discharge	13	0.627924	0.5880	51.37049	0.0000
Max discharge	13	17.6912	0.4259	26.70504	0.0085
Tree cover	13	0.603256	0.3250	17.33055	0.1376
Lagrange-multiplier test					
	Lag	chi2	Df		Prob>Chi2
	1	12.083	16		0.738
	2	18.093	16		0.318
H0: no autocorrelation at lag order					
Johansen tests for cointegration		Number of obs = 36			
		Lags = 3			
Trend: constant					
Sample: 1961 – 1996					
Maximum rank	Parms	LL	Eigenvalue	Trace statistic	5% critical value
0	36	-301.54621		91.5423	47.21
1	43	-278.16672	0.72716	44.7833	29.68
2	48	-264.30566	0.53701	17.0612	15.41
3	51	-257.44184	0.31704	3.3335*	3.76
4	52	-255.77507	0.08844		
Eigenvalue stab. Condition		**Eigenvalue**		**Modulus**	
All the eigenvalues lie inside the unit circle.		0.904		0.904	
VAR satisfies stability conditions.		0.214 -0.7931404i		0.821	
		0.560 -0.5783699i		0.805	
		-0.373 -0.6182924i		0.722	
		-0.694 -0.1734751i		0.715	
		-0.333 -0.2399767i		0.411	
		0.345		0.345	

Granger causality Wald tests				
Equation	**Excluded**	**chi2**	**Df**	**Prob>Chi2**
Average Discharge	Min discharge	16.495	3	0.001
	Max discharge	3.813	3	0.282
	Tree cover	31.702	3	0.000
	ALL	56.789	9	0.000
Min Discharge	Average discharge	1.321	3	0.724
	Max discharge	2.522	3	0.471
	Tree cover	0.753	3	0.861
	ALL	6.809	9	0.657
Max Discharge	Average discharge	10.095	3	0.018
	Min discharge	6.856	3	0.077
	Tree cover	6.794	3	0.079
	ALL	14.229	9	0.114
Tree Cover	Average discharge	1.997	3	0.573
	Min discharge	6.888	3	0.076
	Max discharge	8.914	3	0.030
	ALL	17.266	9	0.045

Based on the VAR (3) lag values, change in tree cover does not significantly influence river flow at a 95% CL with all the variables at lag(3) exhibitng non-significant different P>0.05 as shown (Appendix XI). Overall, the model outputs were contrary to studies that employed SWAT study in the Pacific watershed, Dijo watershed in Ethiopia and Thiba sub-basin, Kenya that suggested a decrease in forest cover results in an increase of peak flow by between 9.3 and 22% and an increase in dry season flow between 8.2 and 60% (Kasuni & Kitheka, 2017; Leta et al., 2018; Nigusie & Dananto, 2021). The findings suggest that tree cover is not the principal predictor of river flow dynamics Ura sub-basins, thus other other bio-geophysical and anthropogenic factors could be exerting pressure on Ura river flows. This is consistent with a couple studies conducted similar ecosystems (Gao et al., 2013; Milliman et al., 2008).

That notwithstanding, the aggregate lagged values for the four variables recorded significance on the average flow when used jointly in a model. This is a suggestion that both the parameters can be a good predictor of average river flow dynamics for Nyambene and similar watersheds. Based on the model estimates and diagnostics, the study generated four structural VAR (3) linear models for the respective variables, in the shown in equation 4.7, 4.8, 4.9 and 4.10. Worth noting the tree cover estimates are different values, thus deployment of the model to consider transforming the final output.

$$\text{Av dis}_{1_t}=5.890_1+[0.20(t-1)+(-0.23)(t-2)+(-0.23)(t-3)]_{\text{av dis}}+[(-0.92)(t-1)+(1.22)(t-2)+(2.74)(t-3)_{\text{min dis}}]+ \quad 4.7$$

$$[(-0.01)(t-1)+(0.05)(t-2)+(-0.04)(t-3)]_{\text{max dis}}+[(2.1)(t-1)+(1.33)(t-2)+(-4.02)(t-3)]_{\text{tree coves}}+2.64_{\text{Av dis}}$$

$$\text{Min dis}_{1_t}=0.17_1+[(0.03)(t-1)+(0.01)(t-2)+(0.0002)(t-3)]_{\text{av dis}}+[(-0.05)(t-1)+(0.43)(t-2)+(0.27)(t-3)_{\text{min dis}}]+ \quad 4.8$$

$$[(0.001)(t-1)+(0.003)(t-2)+(-0.008)(t-3)]_{\text{max dis}}+[(0.14)(t-1)+(0.01)(t-2)+(-0.05)(t-3)]_{\text{tree coves}}+0.52_{\text{min dis}}$$

$$\text{Max dis}_{1_t}=85.80_1+[(-1.40)(t-1)+(-1.59)(t-2)+(-1.2)(t-3)]_{\text{av dis}}+[(3.17)(t-1)+(4.61)(t-2)+(5.59)(t-3)_{\text{min dis}}] \quad 4.9$$

$$+[(0.032)(t-1)+(-0.09)(t-2)+(-0.19)(t-3)]_{max\ dis}+[(9.41)(t-1)+(8.73)(t-2)+(-0.11)(t-3)_{Tree\ cov}]+14.70_{max\ dis}$$

$$\text{Tree cover}_{1_t}=0.52_1+[(0.02)(t-1)+(0.02)(t-2)+(-0.03)(t-3)]_{av\ dis}+[(0.21)(t-1)+(-0.32(t-2)+(-0.16)(t-3)_{min\ dis}]+ \quad 4.10$$

$$[(-0.01)(t-1)+(-0.01)(t-2)+(0.01)(t-3)]_{max\ dis}+[(-0.19)(t-1)+(0.029)(t-2)+(0.17)(t-3)_{Tree\ cov}]+0.50_{tree\ cov}$$

4.4.3.4 Model Test (Ura Discharge)

Using the models, the study predicted and forecast river flow dynamics using a post-estimation command in Stata. The t-test comparison of both the actual means with the respective predicted values exhibited non-significance differences with a p-value> 0.05 for the variables (Table 4.51). Test findings suggest the model-predicted values are not statistically different from the actual values. Equally, the residual mean is roughly zero scores, suggesting that predicted value errors are 'white noise' and thus in compliance with the VAR model assumption (Figure 4.13).

Table 4.51: Summary of Descriptive Statistics on Actual and Predicted Lag Values

Variable	Obs	Mean	Std. Dev.	Min	Max	T-test	
Average Discharge	40	7.134	4.071	2.243	21.64	t=0.0949	df=75,
Predicted Aver discharge	38	7.145	2.03	2.352	12.23	p=0.9246	
Min discharge	40	.944	.865	.002	3.784	t=0.8592,	df=75,
Predicted Min discharge	38	.841	.589	.255	2.487	p=0.3930	
Max Discharge	40	47.53	19.12	9.457	69.24	t=-0.2821,	df=75,
Predicted Max discharge	38	48.05	9.953	21.58	63.60	p=0.7782	
Error	37	0	3.692	-5.413	13.36		

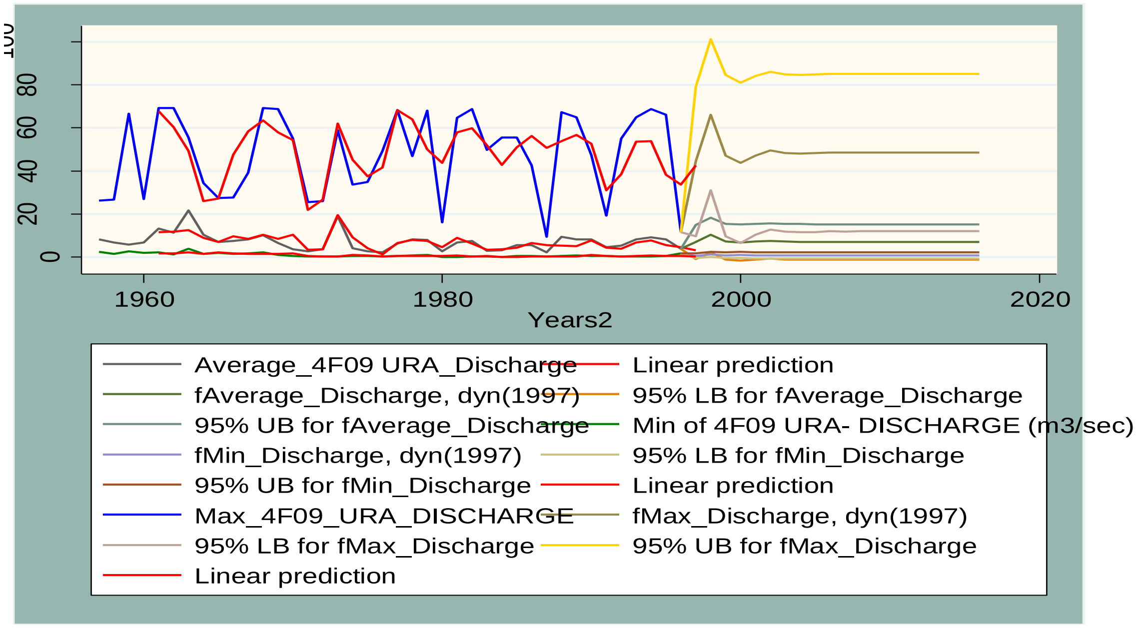

Figure 4.13: Time series line plot for URA (4F09) river flow data

Equally, the test on the tree cover changes impulse response as a function of river flow dynamics, exhibited varied outputs. For instance, the change in unit tree cover slightly increases the average river flows in the first two years before returning to the prevailing steady flow. While a change in a unit tree cover sharply increases, the peak flows in the first two years and a slight decline in the subsequent two years before the subsequent peak flow decline. However, there are no significant changes to base flow with tree cover change over eight years (Figure 4.14Figure). Based on forecast error variance decomposition (FEVD), the change on tree cover would cause up to about1% and 10% change of base flow and peak flow, respectively, as shown (Appendix XIII). The FEVD out put were within range with a SWAT study conducted in Dijo watershed in Ethiopia that reported a peak increase attributed to land cover change to between 9.3% and 10% and a decrease in dry season flow between 8% and 11%(Nigusie & Dananto, 2021). Equally, within range in comparison with a study carried in Thiba which estimated the increase in peak flow attributed to decrease in forest cover to 6% and decrease in dry season flow at about 2% (Kasuni & Kitheka, 2017). However, the study findings were lower compared to a study conducted on a Pacific Island that estimated an increase in peak flows attributed to decline forest cover to approximately 22% and decline dry season flow to up to 60% (Leta et al., 2018).

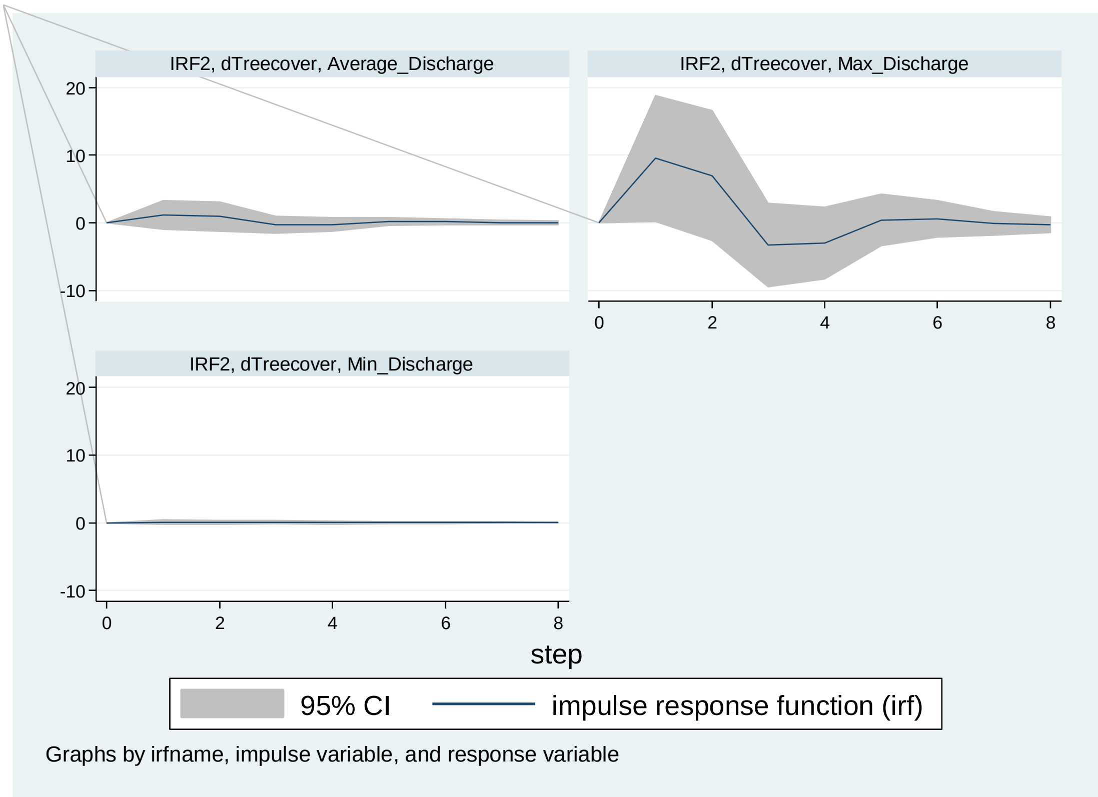

Figure 4.14: Ura (4F09) IRF, impulse (tree cover), response (river flow)

4.4.3.5 Modelling of THAGANTHA Sub-Basin (4F20)

Using the VAR (3) model, analysis of the lagged values on the various parameters exhibited varied responses. The Lagrange multiplier test exhibited non-significance, rejecting the null hypothesis on autocorrelation between the model lags. The Johansen test with four ranks satisfies the condition of no cointegration between lags. This is because a majority of trace statistics recorded higher values than critical values at 95% CL. The test on Eigenvalue generated modulus values less than one a suggestion that all the Eigenvalues line inside the unit circle and thus the VAR (3) model satisfies the stability condition. The Granger causality Walt test exhibited varied output on the influence of the four variables on each other. At a 95% confidence level, the tree cover changes recorded significance on average and base flow with $X^2_{(3)} = 11.292$ $p<0.05$ and $X^2_{(3)} = 21.849$, $p<0.05$, respectively. Equally, tree cover changes recorded significance as a response function to base and peak flow with $X^2_{(3)} = 9.343$ $p<0.05$, and $X^2_{(3)} =13.959$, $p<0.05$, respectively (Table 4.52).

Table 4.52: Thangatha (4F20) VAR (3) Model Statistics

Vector autoregression lag (3)			No. of obs	=	32
Sample: 1969 - 2000			AIC	=	19.39189
			HQIC	=	20.1814
Log-likelihood =		-258.2703	SBIC	=	21.77372
FPE =		3786.816			
Det (Sigma_ml) =		120.3479			

Equation	Parms	RMSE	R-sq	Chi2	P>chi2
Average discharge	13	2.79932	0.5483	38.84054	0.0001
Min discharge	13	1.82225	0.8032	130.5806	0.0000
Max discharge	13	13.1128	0.2914	13.16181	0.3574
Tree cover	13	0.795346	0.6094	49.92329	0.0000

Lagrange-multiplier test

Lag	chi_2	df	Prob>Chi2
1	26.902	16	0.043
2	20.175	16	0.212

H0: no autocorrelation at lag order

Johansen tests for cointegration Number of obs = 33
Lags = 3

Trend: constant

Sample: 1968 – 2000

Maximum rank	**Parms**	**LL**	**eigenvalue**	**Trace statistic**	**Critical value (5%)**
0	21	-275.75156	.	44.6167	29.68
1	26	-260.94833	0.59228	15.0102*	15.41
2	29	-256.42182	0.23992	5.9572	3.76
3	30	-253.44324	0.16516		

Eigenvalue stability condition	**Eigenvalue**	**Modulus**
All the eigenvalues lie inside the unit circle.	0.042 ± 0.9803499i	0.981
VAR satisfies stability conditions.	-0.910	0.910
	0.430 ± 0.7268883i	0.845
	-0.653 ±0.5266138i	0.839
	0.669 ±0.2151474i	0.702
	0.463	0.463
	0.225 ±0.2301771i	0.322

Granger causality Wald tests

Equation	**Excluded**	**chi2**	**df**	**Prob>Chi2**
Average discharge	Min discharge	6.092	3	0.107
	Max discharge	5.473	3	0.140
	Tree cover	11.292	3	0.010
	All	21.459	9	0.011
Min discharge	Aver. discharge	6.697	3	0.082
	Max discharge	6.784	3	0.079
	Tree cover	21.849	3	0.000
	All	40.419	9	0.000
Max discharge	Aver. discharge	0.550	3	0.908
	Min discharge	2.420	3	0.490
	Tree cover	3.874	3	0.275
	ALL	9.103	9	0.428
Tree cover	Aver. discharge	5.276	3	0.153
	Min discharge	9.343	3	0.025
	Max discharge	13.959	3	0.003
	All	49.900	9	0.000

The study findings confirm tree cover change can be an excellent predictor of river flow dynamics and vice versa for the Nyambene and similar watersheds. Further, the aggregate lagged values for the four variables recorded significance on average and base flow when used jointly. This is a suggestion that both parameters are good predictors of base flow. Based on the model estimates and diagnostics, the study generated four structural VAR linear models for the respective variables as shown in the equations 4.11, 4.12, 4.13 and 4.14. Worth noting, the model reports the tree cover outputs are based on the first difference values.

$$\text{Av dis}_{1_t}=4.89_1+[0.2(t-1)+(-0.03)(t-2)+(-0.31)(t-3)]_{\text{av dis}}+[(0.79)(t-1)+(0.03)(t-2)+(0.02)(t-3)_{\text{min dis}}]+ \quad 4.11$$

$$[(0.08)(t-1)+(-0.05)(t-2)+(-0.003)(t-3)]_{\text{max dis}}+[(0.3)(t-1)+(2.07)(t-2)+(-1.87)(t-3)_{\text{Tree cov}}]+2.18_{\text{av dis}}$$

$$\text{Min dis}_{1_t}=-1.77_1+[(0.01)(t-1)+(0.31)(t-2)+(-0.04)(t-3)]_{\text{av dis}}+[(0.95)(t-1)+(-0.55)(t-2)+(0.01)(t-3)_{\text{min dis}}]+ \quad 4.12$$

$$[(0.04)(t-1)+(-0.04)(t-2)+(0.03)(t-3)]_{\text{max dis}}+[(-1.09)(t-1)+(1.58)(t-2)+(-1.1)(t-3)_{\text{Tree cov}}]+1.42_{\text{min dis}}$$

$$\text{Max dis}_{1_t}=27.36_1+[(-0.7)(t-1)+(-0.01)(t-2)+(0.11)(t-3)]_{\text{av dis}}+[(-0.07)(t-1)+(-2.96)(t-2)+(-0.22)(t-3)_{\text{min dis}}]+ \quad 4.13$$

$$[(0.03)(t-1)+(0.09)(t-2)+(-0.14)(t-3)]_{\text{max dis}}+[(-4.99)(t-1)+(-7.5)(t-2)+(-11.32)(t-3)_{\text{Tree cov}}]+10.22_{\text{max dis}}$$

$$\text{Tree cover}_{1_t}=0.01_1+[(-0.06)(t-1)+(-0.02)(t-2)+(0.12)(t-3)]_{\text{av dis}}+[(-0.08)(t-1)+(-0.32)(t-2)+(0.03)(t-3)_{\text{min dis}}] \quad 4.14$$

$$+[(-0.004)(t-1)+(0.01)(t-2)+(-0.04)(t-3)]_{\text{max dis}}+[(-1.14)(t-1)+(-0.92)(t-2)+(-0.61)(t-3)_{\text{Tree cov}}]+0.62_{\text{tree cover}}$$

Model Test

Using the post-estimation command in Stata, the study predicts and forecasted the base flow, average flow, and peak flow for comparison. In that regard, the study ran a t-test between the actual means with the respective predicted values. The analysis exhibited a non-significance difference with an unpaired t-test for both parameters with a p-value>

0.05 (Table 4.53). Test findings suggest the model-predicted values are not statistically different from the actual values. Equally, the residual mean was roughly zero, a suggestion that predicted value errors are 'white noise', thus in compliance with the VAR model assumption.

Table 4.53: Summary of Descriptive Statistics on Actual and predicted lag values

Variable	Obs	Mean	Std. Dev.	Min	Max	T-test
Average discharge	36	6.615	3.435	1.376	15.037	t=0.867, df=67,
Pred aver. discharge	33	5.981	2.521	1.292	12.954	p=0.3893
Min discharge	36	1.94	3.084	.011	11.451	t=-0.114, df=67,
Pred min discharge	33	2.023	2.952	-1.054	11.491	p=0.9096
Max discharge	36	26.224	12.273	7.606	63.28	t=0.231, df=67,
Pred max discharge	33	25.643	7.928	8.2	51.087	p=0.8180
Error	32	0	2.192	-4.972	5.697	

The model's actual, predicted, and presented forecast values are in the time series plot (Figure 4.15). Likewise, the test on tree impulse response as a function of river flow dynamics exhibited varied outputs. For instance, the change in a unit tree cover resulted in slight deep in the average and base river flows in the first two years and a subsequent slight steady increase of flow. While a change in a unit tree cover results in a gentle decline of peak flow in the first year. In the subsequent year, a gradual increase followed and recorded the highest in the fourth year, then a sharp decline before a slightly steady increase between the sixth and eighth year (Figure 4.15Figure). Overall, the model can explain up to 18%, 14%, and 15% of average, base, and peak flows respectively as influenced by tree cover change in the Thangatha basin (Appendix IX).

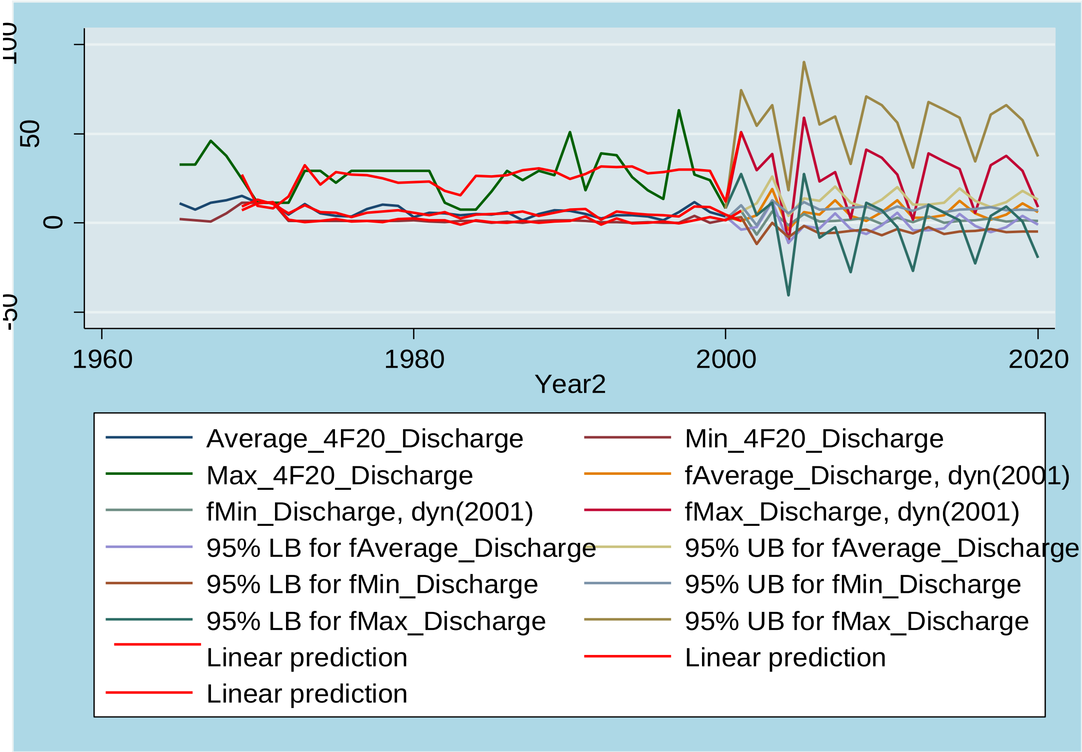

Figure 4.15: Time series line plot for Thangatha (4F20) river flow data

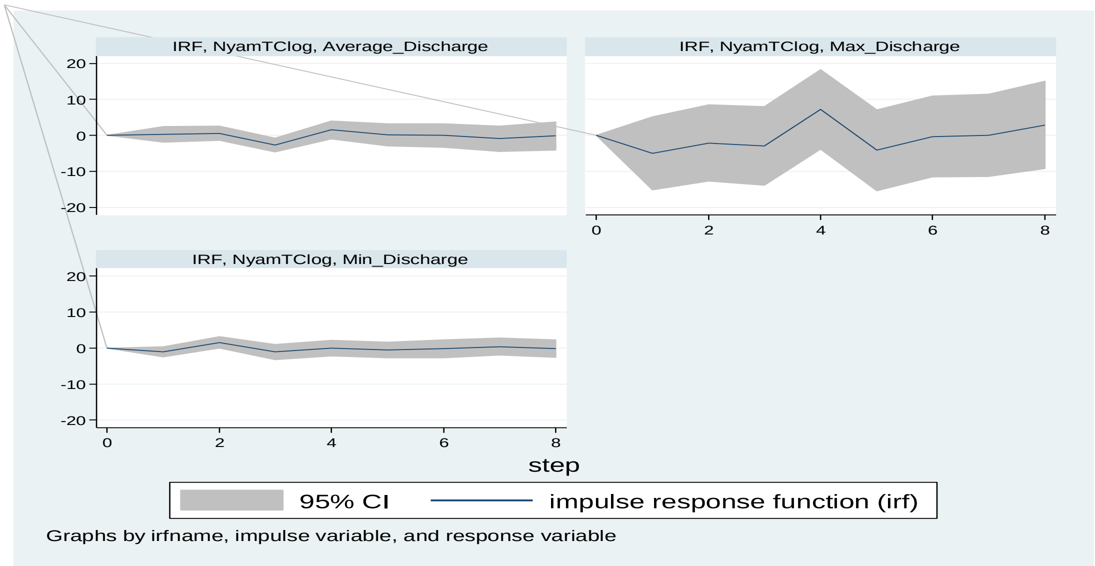

Figure 4.16: Thangatha (4F20) IRF, impulse (tree cover), response (river flow)

In regard, the model confirms that tree cover change influences the base and peak flows as represented by the minimum and max discharge, respectively. The model finding suggests that a change in a unit of tree cover will increase the river base flow by 1.6m^3/sec at a 90% confidence level by two lags. Likewise, a unit change on tree cover reduces the peak flow by 11.3m^3/sec at a 90% confidence level by three lags. The model attributes approximately 61% of the variance in river flow dynamics in the watershed to tree cover change, as shown (Table 4.53). This suggests that tree cover change can be a good predictor of river flow dynamics. The study findings were consistent with most literature suggests that a decrease forest cover increases wet season flow and decreases dry season flows and vice versa (Kasuni & Kitheka, 2017; Leta et al., 2018; Oztürk et al., 2013).

CHAPTER FIVE

CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

5.1.1 Study Findings Summary

The study estimates the total ES value for the two ecosystems at KES 58.8 billion (USD 549.7 million) and KES 39.4 billion (USD 368.4 million) for the Elgeyo and Nyambene WCE, respectively. This translates to KES 542,793.97 (USD 5,072.84) and KES 1,3 million (USD 12,152.99) ha^{-1} $year^{-1}$. The relative proportion placed indirect use values higher at 78% and 70% for the Elgeyo and Nyambene, respectively. Direct use values accounted for 21% and 29% of the total value, respectively, while non-use values accounted for less than one percent of the total for each ecosystem. Disaggregating the total value on a per capita income, it corresponds to between KES 42,416.67 (USD 396.42) and 53,230.77 (USD 497.48) equivalent to 19.4% and 24.4% of Kenya's per capita income. Distribution of indirect use values based on household residing in the two WCE equates to KES 90,042.89 (USD 841.52) and KES 48,803.48 (USD 456.11) HH^{-1} $year^{-1}$, respectively. This translates to between 33% and 35% of the estimated forest community's household average income.

Equally, the study shows that land cover change impacts on stock and flow of ES as exhibited in assessment of forest biomass and river flow dynamics. For instance, the decrease in forest cover per year results in a decline in base flow by between $1mm^3$/sec and $10mm^3$/sec while increasing peak flows to between $16mm^3$/sec and $70mm^3$/sec. Likewise, a unit change in forest species diversity, forest cover, and stem volume attributed land cover change would reduce unit forest biomass by a factor of 1.1, 2.2, and 1.2 on average, respectively.

5.1.2 Assessment of the socioeconomic, perception of ES benefits, forest dependency, and level of exploitation of the Elgeyo and Nyambene water catchments by the communities.

The two ecosystems, like many of the water catchment ecosystems, are critical to household incomes, particularly those bordering forested areas. For instance, direct-use forest products contribute over 30% of the household's income, a confirmation that forest adjacent community heavily relies on forest resources for sustenance and livelihood. However, dependencies vary across geographical location and socioeconomic and cultural attributes. The study has shown that households with lower income (poor households), big household sizes, large-scale herders, and those closer to the ecosystem highly depend on forest resources. In that regard, rejected the null hypothesis and confirms that the sociocultural and economic attributes influence forest dependency. This was also manifest in a test on willingness to pay (WTP) where the community confirmed their willingness to support the conservation of these critical ecosystems regardless of the benefit drawn from them. Notably, though, the willingness to pay is subject to sociocultural and economic factors, thus, the study rejects the hypothesis that willingness to pay is not a function of sociocultural attributes.

5.2.3 Estimating total economic value of ecosystem services for the Elgeyo and Nyambene water catchment ecosystems

In 2021, the Kenyan gross domestic production (GDP) stood at KES 12 trillion, with annual growth of approximately 6% (KES 630 billion) equivalent to KES 218,181.8 (USD 2,040) per capita GDP (World Bank Group, 2021). In that regard, disaggregating the total on a per capita basis, it will range between KES 42,416.67 (USD 396.42) and 53,230.77 (USD 497.48) which corresponds to between 19.4% and 24.4% of the World Bank per capita. These findings imply that the water catchment ecosystems contribute significantly to both local and national GDP besides amelioration of the global climate, such as substantial carbon sequestration. In that regard, the study rejects the hypothesis that such ecosystems don't contribute significantly to the local, national, and global economies. That notwithstanding, the study recorded discrepancies in unit values across the reference literature. It attributes this to ecological dynamics, inevitable product price fluctuations, and the difference in methodological approach and techniques, among others.

5.3.4 Assessing the impact of land use change on the state and flow of ES using biomass and river flow dynamics

The study confirms a significant decline in forest land and records an increase in other land uses, such as cropland within forested watersheds. The change in forest land has resulted in decreased tree cover, which has a significant effect on forest biomass besides loss of biodiversity. Likewise, the study has shown that forests cover change has a significant impact on river flow dynamics, such as a decrease in base and an increase in peak flows as attributed to tree cover decline. Besides other bio-geophysical and social attributes, loss of tree cover contributes to enhanced surface run-off, enhanced peak flows, and reduced base flows besides other. In that regard, the study, therefore, rejects the hypothesis that land cover change and land use change have a non-significant impact on the stock and flow of ecosystem services. Equally, the findings have shown that native forests generate enhanced ecosystem services. Such findings contribute to a debate on the type of forest to adopt, particularly for the restoration and rehabilitation of the water catchment ecosystem. If the unit value of the two ecosystems can be anything to go by, then forested watersheds are better off with native forests as opposed to industrial forests, thus enhancing benefits for humanity.

5.1.5 Modelling the impact of forest cover change on stock and flow of ES using forest biomass and river flow dynamics

The study has shown that the generalised linear model (GLM) adequately quantifies land-based forest biomass while Vector autoregressive (VAR) quantifies multivariate time series data such as river flow dynamics, although at a different level of confidence. The land-based forest biomass model developed by the study has enhanced the stem-based biomass algorithm by incorporating species count, land cover score, and stem count. This model would enable forest and water tower managers to predict future forest biomass particularly in an era of land cover and use change, thus making adequately informed decisions as long as researchers provide them with quoted attributes.

Likewise, the VAR model for river flows reduced the number of variables compared to the traditional river flow prediction model. In that regard, using the previous lagged river flow values, water resource managers can easily predict future river flows, thus better decision-making.

Notably, the assessment of the impact tree cover changes on river flow using the VAR model exhibited varied responses. In that regard, the study reveals that tree cover change can be a good predictor of river flow regimes in a sub-basin with minimal anthropogenic pressure. However, it may not be a good predictor in river systems facing anthropogenic pressure, such as, damming, diversion and over-abstraction, which is the case in most ecosystem in the country (Maingi & Marsh, 2002; Papadaki et al., 2020; Papadaki & Dimitriou, 2021). In that regard, the study fails to reject the null hypothesis.

5.2 Recommendations

- The study acknowledges the gaps, particularly in the valuation of regulatory services. This is largely attributed to data gaps, ecological intricacies, complexities in valuation methods, and inadequate expertise. The shortcomings call for enhanced efforts to improve valuation techniques and approaches to support the realisation of acceptable data. This will go a long way in enhancing the quality of information, thus supporting and facilitating rational decision-making, resource accounting, and conservation policy development.
- The economics of ecosystem service remains marginal, both in local and international economic development discourse and decision-making. Notably, countries have commonly built their prosperity around GDP, which is skewed since it doesn't incorporate all typologies of capital, such as natural capital. This is becoming part of the global contentious discourse, thus the need to develop a robust and integrated tool to measure a country's prosperity. The tool should go beyond the traditional GDP and ensure the incorporation of other ingredients of growth, such as the value of ES, considering that societal decision-making involves trade-offs.

- Considering the low government appropriation and the enormous resource demands, it calls for the establishment of an alternative mechanism to raise resources to support conservation programs. Some of the tested mechanisms include market-based schemes to reward conservation efforts, such as payment of ecosystem services (PES). This will address some of the potential resource users' conflicts and help in designing policy and associated instruments to facilitate and enhance conservation. This will go a long way in facilitating conservation programs, such as rehabilitation, patrols, fencing, and education, among others. However, the establishment of such a scheme should consider establishing a legal framework, sustainable financing mechanism, awareness creation and political goodwill. Without a legal framework, the future of PES would remain uncertain.
- Kenya, like many other developing countries whose population is bearing impact climate change, should pursue their fair share of global conservation kitty, such as REDD+, carbon, and climate action, and now the nationally determined contribution (NDC) emission reduction targets. The initiative should include seeking partnerships with individuals and/or companies with international networks and expertise in carbon funds and markets. Some of these facilities can support sustainable conservation and reduce perennial resource conflict experienced with forest communities around water catchment ecosystems.
- Equally, in the face of land cover and land use change, society has not come to terms and the reality of the impact of such changes in stock and flow of ES, and to that extent, ES accounting and reporting becomes necessary to demonstrate their status and trends. This will facilitate rational decision-making, particularly in promoting policies and management actions that would support sustainable ecosystem conservation.
- Governments should also establish a robust strategy that makes it unattractive to invest in forest extraction businesses. This will reduce water catchment ecosystems' anthropogenic degradation. This is besides enhancing budgetary

allocation to parent ministries to ensure they fully facilitate sustainable conservation of water catchment ecosystems in the country.

- There is also a need to fully enforce the existing legislation, such as the regulation on 10% woodlot establishment on farmland in Kenya. Advocate for the establishment of farm forestry or agroforestry, which goes a long way in not only reducing overreliance and pressure on state forests but improving forest community livelihood as an alternative source of income.
- Governments should design and fully implement natural resource benefit-sharing mechanisms and ensure that the eligible steward's access to resources gained from the sale of marketable goods and services from the respective catchment ecosystems.
- State and non-state actors should enhance efforts in capacity building, particularly community conservation groups, such as CFA, and WRUAs. This will supplement state and non-state actors' efforts in the conservation of water catchment ecosystems in the Country.
- In the era of the burgeoning population and more pressure on state-managed forests, there will be a need to address some inequities among forest communities and enhance poverty alleviation efforts. Society can achieve this through programs such as the diversification of livelihood options, regulated access, and alternative income-generating activities, among others.

5.3 Further Research

Though the study assessed a couple of ecosystem services, some gaps will require further research as follows;

- Albeit the availability of some data on indicator parameters, there still exist enormous gaps in river flow dynamics, precipitation runoff, soil, and sedimentation data, among others. In that regard, the need for a deliberate longitudinal study to monitor and generate information on such key ecosystem indicators. Conservationists can use such data to improve future modelling of the impact of land cover and use change on stock and flow of ecosystem services, thus making a better decision.

- There is also a need to widen the ES scope to include services such as seed dispersal, pest and disease regulation, refugia, genetic pool, and pharmaceutical function, among others.
- Though the study attempted to incorporate insect pollination and microclimate ecosystem value, the study cannot authoritatively link the value of the two services to water catchment ecosystems. In that aspect, there will be a need to undertake a deliberate study aimed at determining the linkages of agricultural farm productivity with forested catchment ecosystem-related pollination and microclimate services.
- As society advances and advocates for schemes such as PES, there will be a need to determine how a beneficiary or the steward can pay and accept respectively for particularly non-marketed ES. This, therefore, calls for a deliberate study to determine comprehensively the maximum willingness to pay and the minimum willingness to accept the non-marketed ecosystem products in the Country.

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APPENDICES

Appendix I: ES Valuation Approaches, Techniques, Data Sources, and References

ES TYPOLOGY (Haines-Young & Potschin-Young, 2018)	TEV FRAMEWORK	Ecosystem Service (s)	Valuation Approach	Valuation Technique	Synthesised Model Explanation	Data Source and References
Biomass (nutrition, fibre, energy, and other materials)	Direct use values	Firewood, fencing/constructions poles, forest honey, timber, thatching grass, game meat, natural medicine	Price based	Market prices	$EV_{CFP}=\sum_{i=1}^{N}((Q_{HH}\times MP_{FP})-C_P)\times HH_B)$ *Where EV is the economic value of consumable forest products; Q is the number of goods extracted per household; MP is the market unit price*	Sourced data on the extraction of forest products and gate prices from the household survey, KII, and local market while it sourced applicable

					of the forest product (in the absence of externalities), and C is the transaction cost (costs of collection, transport, and sale),% is the proportion of households benefit from the ecosystem; HH is the No. of the household around the ecosystem at the time of the survey in 2021.	surrogates and respective prices in cases of distorted data and markets from the literature (Kipkoech et al., 2011; Langat & Cheboiwo, 2010; Langat, 2016; Pascual et al., 2010)
Biomass (nutrition, fibre, energy, and other materials)	Direct use values	Fodder (Grazing/browsing)	Cost based	Avoided Cost	$V_{FGV}=\sum_{i=1}^{N}((DMD_{LC}/\omega_{TLU})\times MP_{Sur})\times HH_{B}$ *Whereby V_{FGV} Value for forest grazing; DMD is the household annual dry matter demand of the respective livestock category; ω is the weight of*	Sourced data on livestock and grazing from the forest from the household, FGD, and KII Local market; Collated

					the surrogate commodity (Hay @ 25kg); MP is the market price of the proxy (Hay @250); HH is the No. of household benefits.	data on dry matter demand from secondary sources (E. H. Jahnke, 1982; OEFFA, 2010; USDA, 2011)
Water	Direct use value	Domestic water	Cost based	Replacement cost	$V_{DW}=\sum_{i=1}^{N}(WD_{HH}/WP_S)\times C_S\times HH_B$ *V is the value for domestic water; WD is household annual domestic water consumed; WP is surrogate water production capacity (11.51m*^{3}*/hr for 10hrs a day); C is the unit cost for the water surrogate (borehole drilling);*	Water consumption data sourced from the household, FGD, and KII Local market; secondary data, including the use of alternative water source (borehole)

					HH is the No. of household benefits.	sourced from literature (Bush, 2009); Unit cost of sinking alternative (Athi Water Services Board, 2015; Ayemba, 2018)
Water	Direct use values	Water for Livestock	Cost based	Replacement cost	$V_{LWD}=\sum_{i=1}^{N}((LWD_{LC}/WP_{Sur})\times C_S)\times HH_B$ *V is the value for livestock water consumption; VWD is the HH annual livestock voluntary water demand; WP is surrogate water production capacity (11.51m^3/hr for 10hrs a day) C is the unit cost for the water surrogate*	No. of livestock and grazing data for the two ecosystems generated through Household, FGD, KII Local market; Data on voluntary water demand collated from secondary

					(borehole drilling and maintenance cost); HH is the No. of household benefits.	sources (IWMI, 2007; Sileshi et al., 2003); alternative water sources data (Bush, 2009); and cost of sinking alternative (borehole)(Athi Water Services Board, 2015; Ayemba, 2018).
Biomass (nutrition, fibre, energy, and other materials)	Direct use values	Value-added from the wood industry and trade	Production based	Factor income	$V_{CFP}=\sum_{i=1}^{N} G_{FP} - C_{VA}$ *Where V is the total value of commercial forest products (wood products for industrial use),* G_{FP} *is the*	Sourced data on forest harvest and beneficiaries from KII, local markets, and local industries other informed

					gross value of the processed products (such as paper, matchsticks, among others.), and C_{VA} is the cost of production.	collated from secondary sources(Langat & Cheboiwo, 2010; Langat et al., 2020).
Physical and experiential / Spiritual and/or emblematic interactions	Direct use values (non-consumptive)	Recreational and Cultural/ Spiritual Values	Stated preference and Revealed preference	Contingency valuation and Travel cost	$V_{NCUV}=\sum_{i=1}^{N} WTP_{NCUV} \times Ni_{HH}$ *V is the value of non-consumptive use values, such as recreational; WTP is the mean maximum willingness to pay for the respective ES; Ni is- the number of households benefiting based on the proportion willing to pay.*	Household interviews to estimate the mean maximum WTP (Eregae et al., 2021) and visitation to the ecosystem; KII and FGD to validate household data.
Mediation of mass flow	Indirect use values	Soil conservation (such as soil loss and	Cost based	Mitigation and avoided	$V_{SC}=\sum LC_A \times SE_{RC} \times C_{Proxy}$	Landsat images analysis using high-resolution

		sedimentation)		cost	*Whereby V_{SC} represents the economic value for forest soil conservation; LC_A is the respective land cover area (ha); SE_{RC} is soil erosion reduction coefficient based per land-cover soil erosion coefficients; C_{proxy} Unit Cost of the Proxy estimated at KES 351 (USD 3.34) per ton of sediment.*	GIS and remote sensing for land cover/use analysis; and secondary data for soil loss coefficients (Adeogun et al., 2016; Hurni, 1988; Kateb et al., 2013; KWTA, 2020b, 2020c; Tessema et al., 2020)
Maintenance of soil formation and composition	Indirect use values	Soil Nutrient Conservation	Cost based	Avoided cost and mitigation cost	$EV_{SNC} = \sum \left[\frac{(S_{LC} \times SNC_{LC})}{(Q_{CF} \times \delta_{CF})} \right] \times P_{CF}$ *Where EV_{SNC} is the economic value of soil conservation; S_{LC} is soil conserved (Kg/ha) of the respective land cover; SNC_{LC} is the soil nutrient (%) content*	Land cover/use data generated from high-resolution Landsat image analysis using

					(N, P, K) in the forest soil; Q_{CF} is the commercial fertiliser applied (kg/acre) estimated at 150kg annually in Kenya; and δ_{CF} is the ratio of commercial fertilisers (51%, NPK-17-17-17); P_{CF} is the unit price of the commercial fertilisers (KES 60/kg)	GIS and remote sensing (KWTA, 2020b, 2020c); Soil sampling and laboratory analysis for soil nutrient/ mineral analysis across the respective land covers and secondary data for soil loss and respective nutrient% and the cost of the surrogate commodity (commercial fertiliser) (Gizaw et al., 2021;

						Okelo et al., 2009; Xi, 2009)
Mediation of Liquid Flow	Indirect use values	Water-flow regulation	Cost based	Avoided cost	$V_{WFR}=\sum_{i=1}^{N} A_{LC} \times P_C \times RR_{Coef} \times C_{Sur}$ *Whereby V_{WFR} Represents the economic value for water flow regulation; A_{LC} represents the area (ha) of the land cover; P_C represents annual precipitation stored/conserved by the ecosystem (equivalent to one less the ratio of annual precipitation runoff to precipitation); RR_{Coef}. Runoff reduction coefficient of the respective land cover (estimated by the precipitation runoff coefficient of the respective land cover/land use subtracted from runoff coefficient of*	Land cover/use analysis (GIS and remote sensing)(KWTA, 2020b, 2020c); Runoff Coefficients across the respective land cover/use (Blume et al., 2007; Goel, 2011; Karamage et al., 2018; Kauffman et al., 2007); surrogate unit cost (artificial

					bare land); C_{prox} represents the unit cost (USD 2.12) to operate and maintain a metric cubic water unit of the surrogate reservoir.	dams) (Eytan & Spuhler, 2020; Nissen-Petersen, 2006; The Ministry of Water and Irrigation & World Bank Kenya, 2005; WRI, 2011).
Mediation of waste, toxic and other nuisance	Indirect use values	Water-quality regulation	Cost based	Avoided and Mitigation cost	$V_{WQ}=Q_{WC}\times\rho$ *Whereby V_{WQ} represents the economic value for ecosystem water quality regulation; Q_{WC} is the quantity of water preserved and purified by the*	Land cover/use analysis by high-resolution GIS and remote sensing (KWTA, 2020b, 2020c); Estimate the precipitation

					ecosystem, represented by the total household consumption; ρ represents the unit cost (USD 0.3/m3) of the surrogate water treatment mechanism.	water conserved by the respective land cover/use in the ecosystem(Jahanifar et al., 2017; Xi, 2009); Unit cost of the surrogate(Fuente et al., 2015)
Atmospheric Composition and climate regulation	Indirect use values	Carbon sequestration (plant and soil)	Price based	Market pricing	$V_{FCR}=\sum_{i=1}^{N} A_{LC}\times Q_C \times \varepsilon_C$ *Whereby V_{FCR} is the economic value for climate regulation, A_{LC} area (ha) of the respective land cover, Q_C is the quantity of carbon dioxide sequestered per unit area by the respective land cover, while $\mathcal{E}_C$ represents the average*	Land cover/ use data (KWTA, 2020b, 2020c); field plant and soil carbon mapping for the respective land cover/use; unit CO2e price (World Bank

					global carbon market price per unit carbon.	Group, 2020)
Atmospheric Composition and climate regulation	Indirect use values	Oxygen generation	Production based	Production function	$V_{O_2} = \sum_{i=1}^{N} CO_{2e_{LC}} \times R_{O_2 CO_2} \times \varepsilon_{O_2}$ Where Vo_2 is the value of oxygen generated by the ecosystem; CO_2e is the carbon dioxide equivalent sequestered by the respective land cover/use; R is the ratio of O_2 to CO_2 (0.73), and Ɛ is the unit cost (USD 2.3/m3) of industrial oxygen production as a surrogate.	CO_2 equivalent estimation using field plant carbon mapping and analysis. Source the concept of oxygen generation through photosynthesis supported by secondary sources (Xi, 2009); while the unit price for the surrogate (industrial oxygen)(Institute

						of Transformative Technologies (ITT) & Oxygen Hub, 2021)
Atmospheric Composition and climate regulation	Indirect use values	Microclimate influence agriculture	Production based	Factor income	$V_{MC}=\sum_{i=1}^{N} Q_C \times \mathcal{E}_C \times R_{MC}$ *V_{MC} is the value of the microclimate influence on agricultural production; Q_C is the annual production/ yields of the respective crops; $\mathcal{E}_C$ is the unit price for the respective crop; R_{MC} is the ratio of ecosystem microclimate influence on crop production.*	Crop yield data (GOK, 2014; MoALF, 2016, 2018; The Republic of Kenya, 1980); crop unit prices (KODI, 2021); Ratio of microclimate influence (Kipkoech et al., 2011)

Life cycle maintenance	Indirect use values	Pollination	Production based	Factor income	$V_{IP}=\sum_{i=1}^{N}(PDR_C \times Q_C \times \mathcal{E}_C)$ Where V_{IP} is the economic value for the ecosystem insect pollination function; PDR_C is the pollination dependency ratio of the respective crop; Q_C is the annual yield (kg) of the respective crop; while $\mathcal{E}_C$ is the market unit price of the respective crop.	Crop yield data(GOK, 2014; MoALF, 2014, 2018; The Republic of Kenya, 1980); crop unit prices(KODI, 2021); Pollination dependency ratio (FAO, 2005; Gallai et al., 2009; Kasina & Kitui, 2007).
	Non-use values	Bequest	Stated Preference	Contingency valuation	$V_{NCUV}=\sum_{i=1}^{N} WTP_{NCUV} \times Ni_{HH}$ *V is the value of non-consumptive use*	Household interviews for the mean maximum WTP (Eregae et al., 2021) and

					values, such as recreational; WTP is the mean maximum willingness to pay for the respective ES; Ni is- the number of households benefiting based on the proportion willing to pay.	visitation to the ecosystem and No. of beneficiaries; KII and FGD to validate household data
Life cycle maintenance, habitat and gene pool protection	Non-use values	Existence/ biodiversity	Stated preference	Contingency valuation	$V_{NCUV} = \sum_{i=1}^{N} WTP_{NCUV} \times Ni_{HH}$ *V is the value of nonconsumptive use values, such as recreational; WTP is the mean maximum willingness to pay for the respective ES; Ni is- the number of households benefiting based on the proportion willing to pay*	Household interviews for the mean maximum WTP(Eregae et al., 2021), visitation to the ecosystem, and No. of beneficiaries; KII and FGD to validate household data

						Ezebilo and Mattsson (2010), Langat 2018 Navrud and Brouwer, 2007; UNEP, 2011, World Bank, 2014, Ruitenbeek, 1989

Appendix II: Household Forest Product Weekly Harvest Statistics

Descriptive Statistics Elgeyo								**Nyambene**					
Ecosystem Services	Measure Unit	Valid% (N=373)	Mean±Std. Dev.	Time ± std. Dev. (hrs)	Trips ± Std Dev.	Domestic Use (%)	Sales (%)	Valid Percent (N=402)	Mean±Std. Dev.	Time ± Std. Dev. (hrs)	Trips ± Std Dev.	Domestic Use (%)	Sales (%)
Fuelwood	B/load	36.9	5.1±5.4	2.7±1.8	2.6±1.9	85.74	14.00	73.88	3.2±2.3	1.3±0.8	1.9±0.8	94.53	6.37
Timber	Fts	3.2	600.00	2.00	6.00	100.00	-	6.47	196.7±471.8	2.6±2.61	1.47±0.72	71.54	28.46
Charcoal	B(90kg)	1.1	500.00	1.00	1.00	-	100.00	11.44	2.47±2.50	1.93±4.20	1.37±0.55	97.83	2.22
Honey	Kg	5.7	129±274.46	2.63±2.13	1.63±0.74	32.14	67.86	12.19	67.23±170.73	1.59±1.11	1.42±0.64	78.91	22.74
Herbs	Kg	12.2	1.00	1.00	1.00	100.00	-	3.48	7.00±8.59	1.29±0.76	1.00	100.00	-
F/Poles	Pcs	7.0	547.1±991	2.9±1.63	1.9±0.9	70.00	30.00	9.45	151±210.7	4.3±5.30	1.5±0.71	97.89	2.11
B/Poles	Pcs	4.0	200±130.9	3.4±1.7	1.00	49.38	50.63	4.23	3.00±1.41	0.7±0.49	1.5±0.7	100.00	-
Wild Fruits	Kg	2.4	1.00	2.00	1.00	100.00	-	37.06	9.44±24.99	0.26±0.47	0.30±0.60	71.51	28.36

Fodder	Kg	5.1	57.4±19.5	1.43±0.5	1.29±0.8	89.29	10.71	20.90	107.6±185.4	1.38±0.7	7.44±8.56	100.00	-
Farm Tools	Pcs	3.3	3.50±1.91	1.38±1.5	1.00	100.00	-	22.89	1.12±0.38	1.12±0.4	1.02±0.2	100.00	-
Marram	Tons	3.0	133±57.7	4.00±1.73	1.00	100.00	-	-	-				
Quarry Stone	Fts							2.49	4,000±2,646	1.83±1.26	1.33±0.58	100.00	-
Mushroom	Kg	9.5	1.38±1.12	1.85±1.46	1.00	92.31	7.69	0.50	55.50±57.28	1.00	1.50±0.71	100.00	-
Reeds	B/load							0.25	20.00	1.00	1.00	100.00	-
Game meat	Kg							0.50	12.00±11.31	2.00±1.41	1.50±0.71	100.00	-
Domestic Water	Ltrs	100.0	626±187	0.50±1.23		100.00	-	100.00	611±285	0.24±0.23	7.00	100.00	-
Distance to the Forest	Km	98.9	3.93±4.88					99.00	1.08±0.98				
Time to the Forest	Mins	99.2	122.7±129.7					100.00	25.70±24.35				
Aggregates		14.9		2.06±1.36	1.71±0.43	78.37	21.61	17.98		1.50±1.41	2.08±1.18	94.15	6.02

Appendix III: Livestock daily water demand and dry matter requirements

Livestock	Mean Weight	TLU	Daily Water (Ltr) Demand Per TLU/Season				DMR
Livestock	Mean Weight	TLU	Wet Season	Dry season (Temp. 15-21°)	Dry Season (Temperature>27°)	Average	
Cattle	300	1.2	14	27	39	26.67	0.025
Goats	25	0.1	20	40	50	36.67	0.038
Sheep	25	0.1	20	40	50	36.67	0.035
Donkey	100	0.4	13	28	40	27.00	0.03
Camel	350	1.4	9	22	31	20.67	0.03
Poultry	2.5	0.01	9	18	32	19.67	0.03
Pig	75	0.3	13	28	40	27.00	0.03
Rabbit	5	0.02	9	18	32	19.67	0.03
TLU	250	1				26.75	0.031

Appendix IV: Total Economic Values and Relative Proportion

Ecosystem Services	Elgeyo		Nyambene	
	Total Value (KES)	**Proportion**	**Total Value (KES)8**	**Proportion (%)**
Fuelwood	3,317,657,267.59	6.75	3,446,443,183.68	9.94
Timber	14,705,292.23	0.03	89,248,580.85	0.26
Charcoal	1,737,430,276.41	3.53	359,725,805.37	1.04
Honey	180,665,018.77	0.37	377,739,756.22	1.09
Natural Medicine	32,960,862.14	0.07	43,565,408.96	0.13
Fencing Poles	430,970,100.00	0.88	19,578,502.24	0.06
Building Poles	140,050,402.14	0.28	648,294.78	0.00
Thatch Grass	-	-	-	-
Wild Fruits	4,577,897.52	0.01	7,347,340.80	0.02
Fodder	73,612,592.12	0.15	1,262,426,578.52	3.64
Grazing Value	2,733,275,359.15	5.56	508,660,570.37	1.47
Farming Tools	245,088.20	0.00	1,253,369.90	0.00
Quarry Stones	-	-	181,522,537.31	0.52
Murram	140,050,402.14	0.28	-	-

Mushroom	49,441,293.22	0.10	86,352,864.18	0.25
Reeds	-	-	25,931,791.04	0.07
Game Meat	-	-	18,670,889.55	0.05
Subtotal	**8,855,641,851.64**	**18.01**	**6,429,115,473.77**	**18.55**
Domestic Water	299,328,576.09	0.61	388,321,059.32	1.12
Livestock Water	568,554,650.04	1.16	224,018,737.93	0.65
Cultural/Spiritual Values	125,686,132.65	0.26	2,388,999,961.33	6.89
Recreational Values	195,602,444.53	0.40	203,840,733.28	0.59
Subtotal	**1,189,171,803.32**	**2.42**	**3,205,180,491.86**	**9.25**
Watershed protection	15,644,985,024.48	31.82	8,671,035,541.16	25.01
Water purification and waste treatment	2,216,402,196.02	4.51	1,228,412,950.51	3.54
Soil Conservation	477,865,598.37	0.97	130,687,341.87	0.38
Soil Nutrient Conservation	149,249,413.90	0.30	40,757,016.20	0.12
CO2 Sequestration (Plant)	2,182,865,005.71	4.44	890,983,229.00	2.57
Soil Carbon Sequestration	16,141,183,055.63	32.83	2,588,952,905.85	7.47
Oxygen generation	830,495,875.90	1.69	397,925,583.83	1.15
Microclimate influence	452,956,020.26	0.92	6,658,365,491.87	19.21
Pollination	821,477,008.61	1.67	4,061,292,428.18	11.72
Subtotal	**38,917,479,198.88**	**79.15**	**24,668,412,488.47**	**71.16**

Bequest	93,616,805.19	0.19	164,098,352.33	0.47
Existence	114,191,927.21	0.23	200,163,924.27	0.58
Subtotal	**207,808,732.40**	**0.42**	**364,262,276.59**	**1.05**
Grand Total	**49,170,101,586.24**	**100.00**	**34,666,970,730.69**	**100.00**

Appendix V: Household annual crop yield and Pollination Dependency Factors

Crop	Pollination Factor	Unit Price	Elgeyo		Nyambene	
			Proportion (N=99,000)	HH Annual Quantity kg)	Proportion (N=173,743)	HH Annual Quantity (Kg)
Avocado	0.65	30.00	0.03	985.71	0.56	611.31
Bananas	-	42.50	0.10	295.86	0.76	0.76
Beans	0.25	89.00	0.04	399.38	0.76	118.21
Butternuts	0.25	103.00			0.12	
Cabbages		14.00	0.22	4,601.06	0.09	493.19
Capsicum	0.05	60.50			0.00	20.00
Carrot	-	44.50	0.01	175.00	0.06	150.00
Chick Peas	0.05	100.50			0.00	
Citrus	0.65	42.50	0.01	80.00	0.08	292.07
Coffee	0.25	187.25	0.03	18.68	0.18	1,212.10
Cotton	0.25	214.00	0.05	300.00	0.00	200.00
Cowpeas	0.25	77.00			0.06	33.91
Dolichos (Njahi)	0.25	150.00			0.10	97.95
French beans	0.05	70.00			0.01	62.50

Green grams	0.25	102.50	0.62	107.43	0.01	38.00
Green Peas	0.25	100.50	0.02	638.33	0.00	4.00
Green pepper	0.05	77.00			0.03	568.10
Guavas	0.25	20.00			0.18	169.49
Irish Potatoes	0.05	60.00	0.23	5,500.19	0.27	208.20
Lentil	0.05	74.55	-		-	-
Linseed	0.05	262.50	-		-	-
Macadamia	0.65	1,449.05	0.23	-	0.53	29.67
Maize	-	28.00	0.80	14,025.10	0.78	1,883.78
Mangoes	0.65	27.00	0.01	87.33	0.05	249.50
Melon	0.25	32.50	0.00	120.00	-	-
Millet		89.00	0.08	900.00	-	
Mirraa (Khat)	-	1,050.00	-	-	0.53	119.46
Onions	0.05	64.50	0.05	1,494.00	0.04	507.00
Pawpaw	0.25	37.00	0.00	100.00	0.02	60.00
Pigeon peas	0.25	220.00	0.00	120.00	0.01	20.00
Pumpkins		30.00	0.01	733.33	0.11	110.88
Pyrethrum		400.00	0.03	450.00	-	
Red Pepper	0.05	77.00	-		0.02	28.33
Soya beans	0.25	52.50	-		0.01	67.50

Sunflower	0.25	31.25	0.01	450.00	-	-
Tea	0.02	201.60	0.02	2,930.00	0.53	1,170.40
Tomatoes	0.05	104.00	0.03	761.82	0.11	179.44
Wheat		33.00	0.12	2,700.00	-	-

Appendix VI: Frequency of Recreational and Spiritual/Cultural Visitation (Nyambene)

Ecosystem Services	Valid Percent (N=402)	Minimum	Maximum	Mean	Std. Error	Std. Deviation
Cultural/Spiritual Forest Use	96.52	0.0	1.0	.567	.0252	.4961
Annual Frequency of Cultural/Spiritual Forest Use	53.98	1	240	32.78	2.477	36.490
Recreational Forest Use	96.27	0.0	1.0	.302	.0234	.4599
Perception of Forest Importance to Cultural/Spiritual Values	99.50	1.0	5.0	3.285	.0644	1.2876
Perception of Forest Importance of the Future Use	99.25	1.0	5.0	3.609	.0477	.9524
Perception of the benefit of the Ecosystem to the Community	96.02	1.0	5.0	2.847	.0597	1.1733
Valid N (listwise)	52.49					

Appendix VII: Household Sampled in the Elgeyo and Nyambene

Elgeyo			Nyambene		
Location	HH	%	Location	Households	Percent
1. Chebior	531	0.54	2. Kiegoi	434	0.25
3. Cheboror	4,517	4.56	4. Ajuki	434	0.25
5. Elgeyo border	266	0.27	6. Akachiu	17,809	10.25
7. Flax	1,594	1.61	8. Amugaa	19,980	11.50
9. Irong	11,427	11.53	10. Antuamburi	869	0.50
11. Kabiemit	3,720	3.75	12. Antuanduru	2,606	1.50
13. Kamogich	4,783	4.83	14. Gitumi	434	0.25
15. Kamwosor	5,049	5.09	16. Kanthiari	9,122	5.25
17. Kapkenda	266	0.27	18. Karama	869	0.50

19. Kapkitony	266	0.27	20. Kiandiu	7,384	4.25
21. Kapkwoni	1,063	1.07	22. Kiegoi	8,687	5.00
23. Kaptagat	9,301	9.38	24. Kigucwa	18,677	10.75
25. Kaptarakwa	3,720	3.75	26. Kimachia	8,687	5.00
27. Kipsaos	1,860	1.88	28. Mbaranga	21,718	12.50
29. Kipsinende	5,580	5.63	30. Miciimikuru	26,061	15.00
31. Kiptulong	4,517	4.56	32. Mikinduri	1,303	0.75
33. Kitany	3,189	3.22	34. Mikinduri East	3,475	2.00
35. Kocholwo	1,594	1.61	36. Mukululu	5,212	3.00
37. Kombatich	266	0.27	38. Nkinyanga	12,162	7.00
39. Maoi	3,720	3.75	40. Thangatha	434	0.25
41. Marichor	6,909	6.97	42. Urru/Mbaranga	7,384	4.25

43. Metkei	1,063	1.07	44. Total	**173,743**	**100.00**
45. Mosop	1,329	1.34	46.		
47. Mutei	7,706	7.77	48.		
49. Nyaru	5,846	5.90	50.		
51. Tembelio	4,252	4.29			
52. Tumeiyo	4,783	4.83			
Total	**99,119**	**100.00**			

Appendix VIII: Soil Mineral Lab Analysis Summaries

KENYA FORESTRY RESEARCH INSTITUTE
Central Highlands Eco-Region Research Programme

Tel: +254-72425978/20202116399
Tel : +254-722157414
Email: cdmuguza@kefri.org
Telegram: "KEFRI" Nairobi
Website: www.kefri.org

P.O. Box 20412
00200 Nairobi
KENYA

Ref: SOILS/10/8/2022 (26)

Date: 3rd August, 2022

SOIL ANALYSIS REPORT

Client: Eregae Ekuwom Justus,
C/o Instititute of Energy & Environmental Technology (IEET) of JKUAT,
P.O Box 62000-00100, Nairobi.
Contacts: 0723805913
Location: Nyambene
Email: jeregae@gmail.com or jeregae@yahoo.com

Lab ID No	Client Reference	Land Cover	Depth (cm)	pH	E.C (mScm)	M.C (%)	B.D (g/cm3)	C (%)	O.M (%)	N (%)	P (ppm)	K (ppm)	Silt (%)
11-14, 17-18, 21-22, 31-32, 39-40	Nyambene 1	Dense forest	Average	4.61	0.12	15.51	1.18	4.94	8.52	0.86	6.23	337.93	19.58
			0-30	4.64	0.13	15.92	0.97	5.23	9.02	1.01	7.87	465.86	18.17
			30-60	4.59	0.11	15.10	1.39	4.65	8.02	0.71	4.60	210.01	21.00
3-6, 15-16, 27-28, 33-38, 41-42	Nyambene 2	Moderate forest	Average	4.70	0.09	13.23	1.19	4.52	7.79	0.91	6.15	410.88	22.25
			0-30	4.56	0.12	12.20	0.99	5.51	9.51	1.05	8.80	562.85	19.75
			30-60	4.84	0.05	14.25	1.40	3.53	6.08	0.76	3.50	258.92	24.75
1-2, 7-10, 19-20, 25-26, 29-30, 43-44	Nyambene 3	Open forest	Average	5.09	0.07	14.97	1.18	4.29	7.40	0.95	6.01	460.05	22.86
			0-30	5.02	0.11	13.30	1.04	5.33	9.19	1.24	7.40	642.77	17.14
			30-60	5.16	0.04	16.64	1.31	3.26	5.62	0.66	4.63	277.32	28.57
23-24, 45-48	Nyambene 4	Other land	Average	5.06	0.15	11.77	1.42	1.60	2.76	0.48	5.72	262.56	24.33
			0-30	4.95	0.03	10.66	1.36	1.93	3.33	0.56	7.40	372.23	23.33
			30-60	5.16	0.28	12.87	1.48	1.27	2.18	0.40	4.03	152.90	25.33
Grand Total				4.84	0.10	14.12	1.21	4.19	7.23	0.85	6.08	388.45	22.02

Compiled by: **Riziki Mwadalu**
Soil Scientist (KEFRI-CHERP)

1

KENYA FORESTRY RESEARCH INSTITUTE
Central Highlands Eco-Region Research Programme

Tel: +254-72425978/20202116399
Tel : +254-722157414
Email: cdmuguza@kefri.org
Telegram: "KEFRI" Nairobi
Website: www.kefri.org

P.O. Box 20412
00200 Nairobi
KENYA

Ref: SOILS/5/1/2021 (40)

Date: 13th January, 2021

SOIL ANALYSIS REPORT

Client: Eregae Ekuwom Justus,
C/o Instititute of Energy & Environmental Technology (IEET) of JKUAT,
P.O Box 62000-00100, Nairobi.
Contacts: 0723805913
Location: Elgeyo
Email: jeregae@gmail.com or jeregae@yahoo.com

Lab ID NO	Client Ref	Land Cover	Depth (cm)	Moisture (%)	pH (H2O)	Ec (mS/cm)	N (%)	Olsen P (ppm)	SOC (%)	SOM (%)	Bulky density (g/cm3)
5/21/101/041, 042, 051, 052, 053, 054, 064, 077, 078, 081, 082	Elgeyo 1	Bushland	Average	62.93	5.77	0.21	0.90	13.13	7.73	13.33	0.94
			0 - 30	62.93	5.74	0.26	0.95	14.54	8.01	13.81	1.10
			30 - 60		5.80	0.17	0.87	11.95	7.50	12.93	0.81
5/21/101/019-020, 023-024, 027-028, 047-048, 055-056, 059-063, 071-072, 089-090, 097-098, 101-102	Elgeyo 2	Degraded forest	Average	59.46	5.69	0.27	0.90	13.60	7.85	13.54	0.93
			0 - 30	59.46	5.66	0.31	0.94	12.06	8.16	14.07	1.06
			30 - 60		5.72	0.22	0.86	15.28	7.52	12.96	0.79
5/21/101/007-010, 033-036	Elgeyo 3	Dense forest	Average	60.33	5.85	0.16	0.89	12.80	7.77	13.39	0.92
			0 - 30	60.33	5.78	0.18	0.95	4.92	7.93	13.66	1.03
			30 - 60		5.92	0.13	0.84	20.67	7.61	13.11	0.81
5/21/101/005-006, 011-014, 067-068, 091-092	Elgeyo 4	Glades	Average	55.47	5.73	0.26	0.90	15.67	7.67	13.23	0.90
			0 - 30	55.47	5.56	0.39	0.95	9.34	8.02	13.84	1.02
			30 - 60		5.90	0.13	0.85	22.01	7.32	12.62	0.78
5/21/101/001-004, 016-018, 031-032, 042-044, 093-	Elgeyo 5	Moderate forest	Average	51.88	5.65	0.26	0.96	12.25	8.26	14.24	0.85
			0 - 30	51.88	5.55	0.31	1.03	9.29	8.83	15.22	0.91
			30 - 60		5.76	0.20	0.88	15.21	7.69	13.26	0.78

1

KENYA FORESTRY RESEARCH INSTITUTE
Central Highlands Eco-Region Research Programme

Tel: +254-72425978/20202116399
Tel : +254-722157414
Email: cdmuguga@kefri.org
Telegram: "KEFRI" Nairobi
Website: www.kefri.org

P.O. Box 20412
00200 Nairobi
KENYA

Ref: SOILS/5/1/2021 (40)

Date: 13th January, 2021

SOIL ANALYSIS REPORT

Client: Eregae Ekuwom Justus,
C/o Instititute of Energy & Environmental Technology (IEET) of JKUAT,
P.O Box 62000-00100, Nairobi.
Contacts: 0723805913
Location: Elgeyo
Email: jeregae@gmail.com or jeregae@yahoo.com

Lab ID NO	Client Ref	Land Cover	Depth (cm)	Moisture (%)	pH (H2O)	Ec (mS/cm)	N (%)	Olsen P (ppm)	SOC (%)	SOM (%)	Bulky density (g/cm3)
094, 099-100											
5/21/101/021-022, 025-026, 045-046, 057-058, 065-066	Elgeyo 6	Other lands e.g. PELIS	Average	53.92	5.45	0.23	0.94	9.44	7.93	13.68	0.92
			0 - 30	53.92	5.53	0.20	0.96	9.80	8.10	13.97	1.04
			30 - 60		5.37	0.26	0.93	9.07	7.77	13.39	0.79
5/21/101/029-030, 037-040, 049-050, 069-070, 073-076, 079-080, 083-088, 095-096	Elgeyo 7	Wooded grassland	Average	58.32	5.64	0.28	0.91	11.84	7.75	13.36	0.90
			0 - 30	58.32	5.54	0.35	0.98	14.31	8.24	14.21	0.98
			30 - 60		5.75	0.21	0.83	9.36	7.26	12.51	0.81
Grand Total				57.47	5.68	0.25	0.91	12.65	7.86	13.56	0.91

Compiled by: **Riziki Mwadalu**
Soil Scientist (KEFRI-CHERP)

Appendix IX: Soil Mineral Across Different Land Covers

Land Cover Category	% Moisture	pH (H2O)	% N (Total)	Olsen P (ppm)	SOC (%)	SOM (%)	Bulky density (g/cm3)
Dense forest	60.33	5.85	0.89	11.86	7.77	13.39	0.90
Moderately dense forest	51.88	5.64	0.96	12.78	8.29	14.29	0.85
Open wooded grassland	58.32	5.64	0.91	11.84	7.75	13.36	0.90
Bushland	63.27	5.76	0.91	12.78	7.80	13.44	0.94
Cropland							
Degraded forest	57.58	5.66	0.91	14.17	7.88	13.59	0.91
Vegetate Wetland							
Others such as PELIS	55.35	5.46	0.94	8.34	7.89	13.61	0.93
Glades	56.99	5.72	0.89	14.57	7.69	13.26	0.94
Open Water Body							
Total	**57.47±8.46**	**5.68±0.37**	**0.91±0.11**	**12.65±10.92**	**7.86±0.8**	**13.56±1.38**	**0.91±0.16**

Appendix X: Moiben 1BA01 VAR (1) Model Statistics

Vector autoregression (1)	No. of obs = 63
Sample: 1956 - 2018	AIC = 10.49527
Log-likelihood = -310.601	HQIC = 10.76286
	SBIC = 11.17563
FPE = 0.425327	
Det (Sigma_ml) = 0.22511	

Equation	Parms	RMSE	R-sq	Chi2	p>chi2
Aver. discharge	5	.737569	0.2073	16.47617	0.0024
Min discharge	5	.093778	0.4115	44.04932	0.0000
Max discharge	5	9.65909	0.1185	8.472711	0.0757
Tree cover	5	1.98586	0.1523	11.31543	0.0232

FPE- Final prediction error AIC- Akaike information criteria HQIC- Hannan Quinn information criteria SBIC- Schwarz's Bayesian information criteria RMSE- Root Mean Square Error

		Coef.	Std. Err.	z	P>z	[95% Conf. Interval]	
Aver.	Average discharge	0.771	0.280	2.760	0.006	0.223	1.319
discharge	Min discharge	-1.510	0.879	-1.720	0.086	-3.233	0.212

	Max discharge	-0.023	0.021	-1.110	0.268	-0.064	0.018
	Tree cover	0.007	0.046	0.160	0.873	-0.082	0.097
	_cons	0.961	0.245	3.930	0.000	0.481	1.441
Min	Average discharge	0.095	0.036	2.670	0.007	0.025	0.165
discharge	Min discharge	0.141	0.112	1.270	0.206	-0.078	0.360
	Max discharge	-0.001	0.003	-0.430	0.670	-0.006	0.004
	Tree cover	-0.014	0.006	-2.380	0.017	-0.025	-0.002
	_cons	0.104	0.031	3.350	0.001	0.043	0.165
Max	Average discharge	8.309	3.662	2.270	0.023	1.132	15.486
discharge	Min discharge	-19.150	11.508	-1.660	0.096	-41.71	3.406
	Max discharge	-0.431	0.275	-1.570	0.117	-0.971	0.109
	Tree cover	0.977	0.600	1.630	0.103	-0.198	2.152
	_cons	10.816	3.205	3.370	0.001	4.534	17.099
Tree cover	Average discharge	-1.648	0.753	-2.190	0.029	-3.123	-0.172
	Min discharge	-2.541	2.366	-1.070	0.283	-7.178	2.096
	Max discharge	0.103	0.057	1.820	0.069	-0.008	0.214
	Tree cover	-0.006	0.123	-0.050	0.962	-0.247	0.236
	_cons	0.460	0.659	0.700	0.485	-0.831	1.752

Appendix XI: Ura (4F09) VAR (3) Model Statistics

Vector autoregression lag (3)		No. of obs.	= 36
Sample: 1961 - 1996		AIC	= 17.09861
Log-likelihood = -255.7751		HQIC	= 17.89695
		SBIC	= 19.38592
FPE = 359.0717			
Det (Sigma_ml) = 17.43043			

Equation	Parms	RMSE	R-sq	Chi2	p>chi2
Aver discharge	13	3.18243	0.6378	63.40037	0.0000
Min discharge	13	0.627924	0.5880	51.37049	0.0000
Max discharge	13	17.6912	0.4259	26.70504	0.0085
Tree cover	13	0.603256	0.3250	17.33055	0.1376

		Coef.	Std. Err.	z	P>z	[95% Conf. Interval]	
Average	Aver. discharge	0.202	0.163	1.240	0.216	-0.118	0.522
discharge	Aver. discharge	-0.227	0.145	-1.560	0.119	-0.512	0.058
	Aver. discharge	-0.226	0.155	-1.450	0.146	-0.530	0.078
	Min discharge	-0.916	1.096	-0.840	0.403	-3.064	1.232

	Min discharge	1.215	0.785	1.550	0.122	-0.324	2.754
	Min discharge	2.736	0.973	2.810	0.005	0.828	4.644
	Max discharge	-0.011	0.032	-0.350	0.723	-0.074	0.051
	Max discharge	0.047	0.031	1.510	0.130	-0.014	0.107
	Max discharge	-0.037	0.028	-1.310	0.189	-0.092	0.018
	Tree cover	2.100	0.881	2.380	0.017	0.373	3.828
	Tree cover	1.332	0.814	1.640	0.102	-0.264	2.928
	Tree cover	-4.024	0.822	-4.900	0.000	-5.636	-2.413
	Constant	5.890	2.644	2.230	0.026	0.708	11.071
Min discharge	Aver. discharge	0.031	0.032	0.970	0.331	-0.032	0.094
	Aver. discharge	0.010	0.029	0.360	0.717	-0.046	0.067
	Aver. discharge	0.000	0.031	0.000	1.000	-0.060	0.060
	Min discharge	-0.052	0.216	-0.240	0.811	-0.475	0.372
	Min discharge	0.425	0.155	2.740	0.006	0.121	0.729
	Min discharge	0.274	0.192	1.430	0.154	-0.103	0.650
	Max discharge	0.001	0.006	0.110	0.912	-0.012	0.013
	Max discharge	0.003	0.006	0.540	0.587	-0.009	0.015
	Max discharge	-0.008	0.006	-1.510	0.130	-0.019	0.002
	Tree cover	0.142	0.174	0.820	0.413	-0.198	0.483
	Tree cover	0.015	0.161	0.090	0.927	-0.300	0.330

	Tree cover	-0.054	0.162	-0.340	0.737	-0.372	0.263
	_cons	0.171	0.522	0.330	0.743	-0.851	1.193
Max discharge	Aver. discharge	-1.403	0.907	-1.550	0.122	-3.182	0.376
	Aver. discharge	-1.594	0.808	-1.970	0.049	-3.178	-0.010
	Aver. discharge	-1.199	0.862	-1.390	0.164	-2.889	0.490
	Min discharge	3.169	6.092	0.520	0.603	-8.771	15.110
	Min discharge	4.612	4.365	1.060	0.291	-3.944	13.167
	Min discharge	5.589	5.411	1.030	0.302	-5.016	16.194
	Max discharge	0.032	0.178	0.180	0.859	-0.317	0.380
	Max discharge	-0.090	0.172	-0.520	0.600	-0.427	0.247
	Max discharge	-0.193	0.156	-1.230	0.218	-0.499	0.114
	Tree cover	9.410	4.899	1.920	0.055	-0.192	19.013
	Tree cover	8.731	4.526	1.930	0.054	-0.139	17.601
	Tree cover	-0.106	4.570	-0.020	0.982	-9.063	8.852
	_cons	85.805	14.697	5.840	0.000	57.000	114.610
Tree cover	Aver. discharge	0.016	0.031	0.520	0.606	-0.045	0.077
	Aver. discharge	0.020	0.028	0.730	0.466	-0.034	0.074
	Aver. discharge	-0.028	0.029	-0.960	0.335	-0.086	0.029
	Min discharge	0.205	0.208	0.990	0.324	-0.202	0.612
	Min discharge	-0.319	0.149	-2.140	0.032	-0.610	-0.027

Min discharge	-0.158	0.185	-0.860	0.390	-0.520	0.203
Max discharge	-0.011	0.006	-1.810	0.070	-0.023	0.001
Max discharge	-0.010	0.006	-1.780	0.075	-0.022	0.001
Max discharge	0.008	0.005	1.460	0.144	-0.003	0.018
Tree cover	-0.185	0.167	-1.110	0.267	-0.513	0.142
Tree cover	0.029	0.154	0.190	0.851	-0.274	0.331
Tree cover	0.175	0.156	1.120	0.262	-0.130	0.480
_cons	0.527	0.501	1.050	0.293	-0.455	1.510

Appendix XII: Thananthu (4F20) VAR (3) Model Statistics

Vector autoregression lag (3)	No. of obs =	32
Sample: 1969 - 2000	AIC =	19.39189
	HQIC =	20.1814
Log-likelihood = -258.2703	SBIC = 21.77372	
FPE = 3786.816		
Det (Sigma_ml) = 120.3479		

Equation	Parms	RMSE	R-sq	Chi2	P>chi2
Average discharge	13	2.79932	0.5483	38.84054	0.0001
Min discharge	13	1.82225	0.8032	130.5806	0.0000
Max discharge	13	13.1128	0.2914	13.16181	0.3574
Tree cover	13	0.795346	0.6094	49.92329	0.0000

		Coef.	Std. Err.	z	P>z	[95% Conf.	Interval]
Average discharge	Aver. discharge	0.204	0.259	0.790	0.431	-0.304	0.712
	Aver. discharge	-0.027	0.201	-0.130	0.893	-0.421	0.367
	Aver. discharge	-0.310	0.248	-1.250	0.212	-0.797	0.177
	Min discharge	0.785	0.404	1.940	0.052	-0.006	1.576
	Min discharge	0.028	0.440	0.060	0.949	-0.834	0.890
	Min discharge	0.021	0.323	0.060	0.948	-0.611	0.653
	Max discharge	0.078	0.039	1.990	0.047	0.001	0.155
	Max discharge	-0.050	0.040	-1.250	0.211	-0.128	0.028
	Max discharge	-0.003	0.039	-0.080	0.936	-0.079	0.073
	Tree cover	0.299	1.109	0.270	0.787	-1.875	2.473

	Tree cover	2.072	1.296	1.600	0.110	-0.468	4.611
	Tree cover	-1.874	1.240	-1.510	0.131	-4.304	0.555
	Constant	4.890	2.181	2.240	0.025	0.615	9.166
Min discharge	Aver. discharge	0.013	0.169	0.080	0.938	-0.318	0.344
	Aver. discharge	0.307	0.131	2.350	0.019	0.051	0.564
	Aver. discharge	-0.044	0.162	-0.270	0.786	-0.361	0.273
	Min discharge	0.953	0.263	3.630	0.000	0.438	1.469
	Min discharge	-0.549	0.286	-1.920	0.055	-1.110	0.012
	Min discharge	0.010	0.210	0.050	0.960	-0.401	0.422
	Max discharge	0.042	0.026	1.660	0.097	-0.008	0.093
	Max discharge	-0.041	0.026	-1.570	0.116	-0.092	0.010
	Max discharge	0.030	0.025	1.190	0.233	-0.019	0.079
	Tree cover	-1.094	0.722	-1.520	0.130	-2.509	0.321
	Tree cover	1.580	0.843	1.870	0.061	-0.073	3.233
	Tree cover	-1.096	0.807	-1.360	0.175	-2.677	0.486
	_cons	-1.772	1.420	-1.250	0.212	-4.556	1.011
Max discharge	Aver. discharge	-0.700	1.214	-0.580	0.564	-3.079	1.678
	Aver. discharge	-0.006	0.941	-0.010	0.995	-1.850	1.837
	Aver discharge	0.112	1.163	0.100	0.924	-2.169	2.392
	Min discharge	-0.070	1.891	-0.040	0.970	-3.777	3.636
	Min discharge	-2.956	2.060	-1.430	0.151	-6.994	1.082
	Min discharge	-0.222	1.511	-0.150	0.883	-3.184	2.739
	Max discharge	0.033	0.184	0.180	0.856	-0.327	0.394
	Max discharge	0.088	0.187	0.470	0.636	-0.278	0.455
	Max discharge	-0.136	0.181	-0.750	0.452	-0.490	0.218
	Tree cover	-4.994	5.195	-0.960	0.336	-15.176	5.188
	Tree cover	-7.498	6.069	-1.240	0.217	-19.392	4.397
	Tree cover	-11.32	5.807	-1.950	0.051	-22.698	0.065

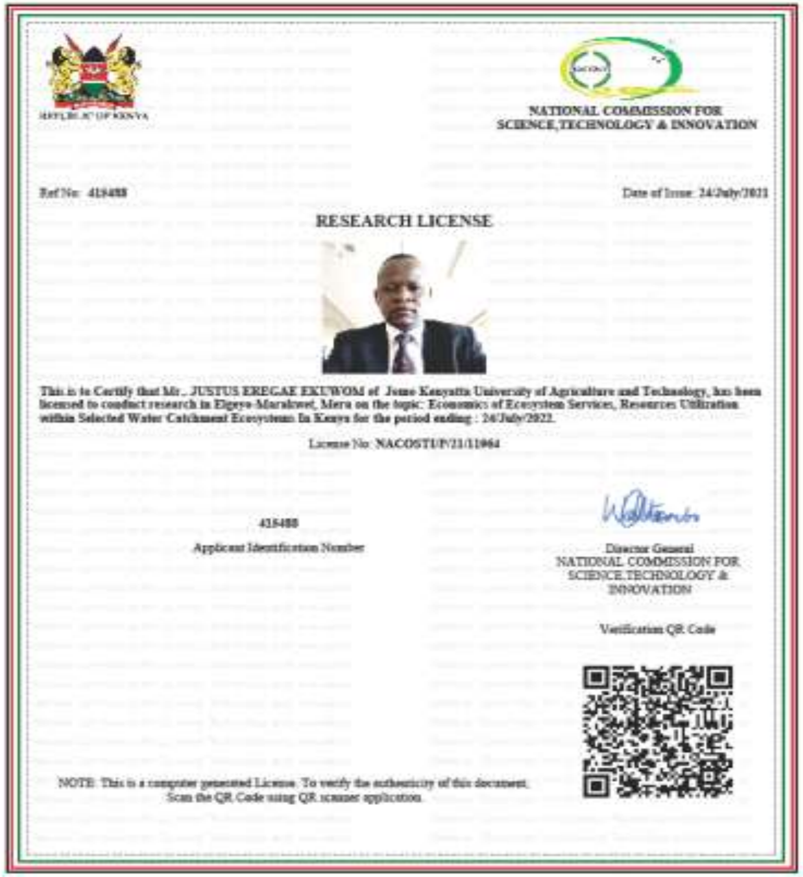
	_cons	27.360	10.218	2.680	0.007	7.332	47.388
Tree cover	Aver. discharge	-0.056	0.074	-0.760	0.447	-0.200	0.088
	Aver. discharge	-0.020	0.057	-0.350	0.725	-0.132	0.092
	Aver. discharge	0.106	0.071	1.510	0.131	-0.032	0.245
	Min discharge	-0.080	0.115	-0.700	0.485	-0.305	0.145
	Min discharge	-0.318	0.125	-2.550	0.011	-0.563	-0.073
	Min discharge	0.026	0.092	0.280	0.778	-0.154	0.205
	Max discharge	-0.004	0.011	-0.350	0.723	-0.026	0.018
	Max discharge	0.012	0.011	1.020	0.308	-0.011	0.034
	Max discharge	-0.039	0.011	-3.590	0.000	-0.061	-0.018
	Tree cover	-1.140	0.315	-3.620	0.000	-1.757	-0.522
	Tree cover	-0.915	0.368	-2.490	0.013	-1.637	-0.194
	Tree cover	-0.611	0.352	-1.730	0.083	-1.301	0.080
	_cons	0.011	0.620	0.020	0.986	-1.204	1.226

Appendix XIII: Model IRF Results (Moiben, Ura, and Thananthu) Impulse tree cover, while response average min and max discharge

IRF Moiben (1BA01)	(1)	(2)	(3)
step	Fevd	fevd	fevd
0	0	0	0
1	0	0	0
2	0.000	0.055	0.033
3	0.000	0.049	0.033
4	0.001	0.047	0.035
5	0.001	0.047	0.035
6	0.001	0.047	0.035
7	0.001	0.047	0.035
8	0.001	0.047	0.035
Ura (4F09)			
1	0	0	0
2	0.020	0.006	0.075
3	0.031	0.006	0.095
4	0.032	0.010	0.100
5	0.032	0.010	0.104
6	0.033	0.010	0.104
7	0.033	0.010	0.103

8	0.033	0.010	0.103
Thangatha (4F20)			
0	0	0	0
1	0	0	0
2	0.003	0.073	0.064
3	0.013	0.145	0.065
4	0.189	0.162	0.077
5	0.217	0.149	0.160
6	0.205	0.144	0.172
7	0.177	0.139	0.160
8	0.176	0.139	0.145
95% lower and upper bounds reported (1) IRF, impulse = Tree Cover, and response = Average Discharge (2) IRF, impulse = Tree Cover, and response = Min Discharge (3) IRF, impulse = Tree Cover, and response = Max Discharge			

Appendix XIV: Research Permits



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Date of Issue: 24/July/2021

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This is to Certify that Mr.. JUSTUS EREGAE EKUWOM of Jomo Kenyatta University of Agriculture and Technology, has been licensed to conduct research in Elgeyo-Marakwet, Meru on the topic: Economics of Ecosystem Services, Resources Utilization within Selected Water Catchment Ecosystems In Kenya for the period ending : 24/July/2022.

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KENYA Forest Service

Kenya Forest Service Hqs
Karura, Off Kiambu Rd
P. O. Box 30513 - 00100
Nairobi, Kenya

Ref. No. RESEA/1/KFS/VOL.V (25)

Date: 13th November 2019

Justus Eregae Ekuwom
P. O. Box 42903-00100,
NAIROBI

RE: PERMISSION TO UNDERTAKE ACADEMIC STUDY AT NYAMBENE AND ELGEYO FOREST.

Reference is made to your letter dated 6th September 2019 in which you requested to conduct research on the *Economics of Ecosystem Services, Resource Utilization and Impact on stock and flow of benefits from water catchment ecosystems in Kenya.*

Permission is hereby granted for you to carry out your academic research in Nyambene and Elgeyo forests.

This permit is valid from 18th November 2019 to 30th November 2020. As part of the requirements, a report must be availed to the Chief Conservator of Forests on completion of your work.

By a copy of this permit, the respective Ecosystem Conservators are hereby instructed to facilitate unhindered access.

Julius Kamau
Chief Conservator of Forests

Copy to: Ecosystem Conservators-Meru and Elgeyo Marakwet.

Trees for better lives

Tel: (254)020-3754904/5/6, (254)020-2014663, (254)020-2020285, Fax: (254)020-2385374
Email: info@kenyaforestservice.org, Website: www.kenyaforestservice.org