Ethiopian Institute of Architecture, Building Construction and City Development (EiABC)

Quantification and Mapping of Ecosystem Services for the Conservation and Management of Dire and Legedadi watersheds, Central Highlands of Ethiopia

Simeneh Admasu Namaga

A Dissertation Submitted to the Ethiopian Institute of Architecture, Building Construction, and City Development in Fulfillment of the Requirements for the Degree of Doctor of Philosophy in Environmental Planning

December 2023

Addis Ababa University

School of Graduate Studies

This is to certify that the dissertation submitted by Simeneh Admasu Namaga entitled **"Quantification and Mapping of Ecosystem Services for the Conservation and Management of Dire and Legedadi Watersheds, Central Highlands of Ethiopia"** in Fulfillment of the Requirements for the Degree of Doctor of Philosophy in Environmental Planning complies with the regulations of the university and meets the accepted standards concerning to originality and quality.

Signed by the Examining Committee:

Responsibility	Name	Signature	Date
Internal examiner	Dr. Aramde Fetene		27 December 2023
External examiner	Professor Belay Simane		27 December 2023
Principal supervisor	Professor Kumelachew Yeshitela		27 December 2023
Co-supervisor	Professor Mekuria Argaw		27 December 2023
Chairman	Dr. Webshet Berhanu		27 December 2023

Dedication

This work is dedicated to all nature conservationists worldwide.

Statement of Author

Unless otherwise referenced, the work provided in this thesis is the researcher’s work and has not been submitted elsewhere for any other degree or qualification.

Student's name: Simeneh Admasu Namaga

Signature: ______________

December 2023

Acknowledgments

I write this note of acknowledgment in a state of gratitude to God for giving me the strength to complete my doctoral study. I express my deepest gratitude to my advisor, Professor Kumelachew Yeshitela for his supervision and knowledge transfer. The comments and feedback on the proposal and dissertation were not only crucial to the completion of this study but also to educational courses, and for this, I am deeply grateful. I'm extremely fortunate and honored to have Professor Mekuria Argaw as my co-supervisor. I appreciate all his unwavering support, guidance, scholarly and strong comments, and constructive suggestions. I value all the knowledge and wisdom both supervisors have shared with me.

I would like to acknowledge the Addis Ababa Water and Sewage Authority (AAWSA), National Meteorological Agency (NMA), and Bereh Woreda Agriculture and Natural Resource Protection Office for providing relevant data for my research work. I am also grateful to Professor Hailu Worku, Dr. Aramde Fetene, Dr. Hayal Desta, Dr. Asfaw Mohammed, Dr. Mesfin Sahle, Dr. Admasu Shibru, Meseret Habtamu, Gizaw Ibissa, and Dr. Hamere Yohannnes for the support provided in various capacities. Work colleagues Lakew Berhanu, Yilma Delelegn, Tola Torkoba, Dr. Mekbeb Tessema, Dr. Zelealem Tefera, and others thank you all.

My family members were always with me during this work. My lovely wife Tsion Fikadu, Genet Abdella, Tirunesh Namaga, Dr. Nega Namaga, Anteneh Sisay, and Dr. Hayatu Awel; thank you all, you deserve big recognition. All the anonymous reviewers and editors of the journals are also thanked for their critical comments that improved all the manuscripts. Finally, my acknowledgment goes to the ILM-ESAW thematic research project of Addis Ababa University; for granting a research fund for my research work. Above all, my gratitude and thanks to God, for being with me from the beginning to the end of everything I did.

Contents **Page**

Table

Figure

Acronym

AAWSA	Addis Ababa Water and Sewerage Authority
CS	Coefficient of Sensitivity
EBWSM	Ecosystem-Based Water Supply Management
ES	Ecosystem Service
ESV	Ecosystem Service Valuation
FAO	Food and Agriculture Organization
GDP	Gross Domestic Product
GIS	Geographic Information Systems
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs
LULC	Land Use Land Cover
MASL	Meter Above Sea Levels
MCM	Million Cubic Metres
MEA	Millennium Ecosystem Assessment
OLI	Operational Land Imager
PES	Payment for Ecosystem Services
PWS	Payments for Watershed Services
RS	Remote Sensing
TEEB	The Economics of Ecosystems and Biodiversity
TIRS	Thermal Infrared Sensor
USGS	United States Geological Survey
UTM	Universal Transverse Mercator

Publications

I. **Simeneh Admasu**, Kumelachew Yeshitela, and Mekuria Argaw (2023a). **Impact of Land Use and Land Cover Changes on Ecosystem Services Values in the Dire and Legedadi Watersheds, central highland of Ethiopia**. Heliyon 9:4.

II. **Simeneh Admasu**, Kumelachew Yeshitela, and Mekuria Argaw (2023b). **Assessing habitat quality using the InVEST model in the Dire and Legedadi watersheds, central highland of Ethiopia**. Sustainable Environment 2765-8511. DOI 10.1080/27658511.2023.2242137.

III. **Simeneh Admasu**, Kumelachew Yeshitela, and Mekuria Argaw (2023c). **Impacts of land use land cover changes and climate variability on water yield in the central watersheds of Ethiopia**. Water Conservation Science and Engineering, 8:14.

IV. **Simeneh Admasu**, Hayal Desta, Kumelachew Yeshitela, and Mekuria Argaw (2022). **Analysis of Land Suitability for Apple-Based Agroforestry Farming in Dire and Legedadi Watersheds of Ethiopia: Implication for Ecosystem Services**. Heliyon 8 (2022) e11217.

V. **Simeneh Admasu**, Kumelachew Yeshitela and Mekuria Argaw (2023d). **Planning for ecosystem services compensation in the central highlands of Ethiopia.** Integrated Environmental Assessment and Management, DOI: 10.1002/ieam.48

Submitted

VI. **Simeneh Admasu**, Kumelachew Yeshitela, and Mekuria Argaw (2023e). **Recreational potential assessment in the Dire and Legedadi landscapes: Implication for rural ecotourism development**. Under review

Quantification and Mapping of Ecosystem Services for the Conservation and Management of Dire and Legedadi watersheds, Central Highlands of Ethiopia

Simeneh Admasu Namaga

Abstract

Quantification and mapping of ecosystem service is a critical endevours for the management of the ecosystems and integrating ecosystem service in land use management and planning. The study assessed the impact of Land Use Land Cover (LULC) dynamics on Ecosystem Service in the Dire and Legedadi watersheds. First, the benefits transfer method was applied to evaluate the ecosystem service value (ESV) changes in response to LULC. Second, the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) model was used to determine habitat quality and assess the watersheds' annual water yield capacity. Thirdly, the previously developed recreational indicators in the ArcGIS tool were also applied to assess recreational potential. Fourth, land suitability for apple farming was assessed using the FAO land evaluation methodology and finally, the premises of compensation for ecosystem services was assessed as a watershed management and planning tool.

The findings revealed that both watersheds experienced considerable LULC changes between 1985 and 2022. Natural vegetation, grassland, and eucalyptus plantations declined dramatically as settlement and cultivation increased. As a result, both watersheds experienced a substantial decrease in ESV and habitat quality. Total ESV in the Legedadi watershed has decreased from approximately US$ 65.8 million in 1985 to approximately US$ 11.9 million in 2022, and from approximately US$ 42.7 million in 1985 to approximately US$ 9.66 million in 2022, respectively. Total ESV in the Dire watershed decreased from approximately US$ 437 thousand in 1985 to approximately US$ 59 thousand in 2022, and from

approximately US$ 225 thousand in 1985 to approximately US$ 36 thousand in 2022, according to global and local ESV estimates.

The area of high habitat quality habitat in the Legadadi watershed has shrunk from 206 km^2 in 1985 to 50.26 km^2 in 2022. Similarly, high habitat quality habitat has gradually declined in the Dire watershed, from 87.29 km^2 in 1985 to 35.44 km^2 in 2022. In terms of water-yielding capacity, the watershed's total water yield increased between 1995 and 2021. The increase in water yield was greatly associated with increased rainfall and in the Legedadi watershed total water yield increased by 15.32%, while in the Dire watershed total water yield increased by 32.5%. Climate variability has had a greater impact on annual water yield than land use changes. The watersheds possess considerable potential for outdoor recreation, with approximately 19% and 23% of the Legedadi and Dire landscapes, respectively, exhibiting supreme recreational potential. Further, considerable land is highly suitable for apple farming, about 6.7%, and 13.1% in the Legedadi and Dire watersheds respectively.

The conversion of land into other economic land uses could potentially affect the sustainable ecosystem production capacity of the watersheds. Landscape restoration integrated with a sustainable agricultural development approach would ensure the sustainability of both agricultural production and ecosystem service synergies without negatively affecting biodiversity. Therefore, the study recommends that designing market-based innovative mechanisms is critical to ensure the active participation of relevant stakeholders, particularly smallholder farmers.

Keywords: LULC, Ecosystem, Watershed, ESV, Habitat quality, Water yield, Recreation, Apple, Ecosystem market

Chapter One

1 Background

1.1 Introduction

The ecosystem provides a diverse range of ecosystem benefits that are required for humans (Costanza *et al.,* 1997; Pereira *et al.,* 2005; Yohannes *et al.*, 2012; Willemen *et al.,* 2015; Stave *et al.*, 2017; TEEB, 2018; Vargas *et al.*, 2019). Ecosystem services provision depends on the state the of natural environment (Summers *et al.*, 2012; Maes and Jacobs, 2017; Willemen *et al.,* 2017). The ecosystem functionality and process rely on the intactness of the natural habitat and are considerably affected by the proximity to adverse land uses and the intensity of land uses (Fu *et al.,* 2013; Yan *et al.,* 2018; Hamere et al., 2021). Key biodiversity areas those rich in species (Duarte et al., 2016), greater endemism (Myers et al., 2000; Naidoo et al., 2006; McBride et al., 2007), and a well-connected and conserved landscapes (Ribeiro et al., 2011; Choi et al., 2018) are more intact ecosystems, and lower species and ecosystem extinction risk (Cardinale et al., 2012; Harrison et al., 2014). The integrity and intactness of an ecosystem and its ecological processes, particularly its natural state, determine its completeness and functionality (Johnson, 2007; Oliver et al., 2015). The loss of ecosystem integrity and intactness reduces the quality of native biota habitat, disrupts ecological processes, and functions, and reduces ecosystem resilience and services (Johnson, 2007; Oliver et al., 2015).

Rapid population growth and environmental and social changes are major causes of global biodiversity decline and affect ecosystem processes (Brooks *et al.*, 2006). The degradation of the natural environment has become a prime ecosystem threat (Haddad *et al.,* 2015). Environmental pressures including overgrazing, and the spread of invasive species affect ecosystem conditions and services (MEA, 2005; Pereira *et al.*, 2005). Policy and decision-makers are requiring detailed spatially explicit indicators and tools to understand the effects of land degradation on the ecosystem (de Groot *et al.,* 2010).

Globally, it is very difficult to secure the required financial support for the conservation of important biodiversity landscapes that do not demonstrate societal benefits (Robertson and Wunder, 2005; Goldman et al., 2008; Duarte et al., 2016). Valuation of ecosystem services could enhance biodiversity conservation by providing further justification for appropriate nature conservation, as well as increased financial support (Goldman et al., 2008; Goldman and Tallis, 2009). Integrating development strategies could help to improve ecosystem trade-offs (Faleiro and Loyola, 2013). In this regard, the approach has the potential to protect areas outside of legally protected reserves (MEA, 2005; Polasky et al., 2005; Goldman et al., 2008; Wendland et al., 2010).

Ethiopia is experiencing a dramatic era of development; the high cumulative demand for socio-economic necessities driven by the invariably fast-growing population is raising the necessity for rapid economic transformation. Most development plans have been focused on short-term economic feasibility to maximize economic outcomes while paying little attention to their environmental impacts, resulting in environmental deterioration (Adugna, 2016). The rapid expansion of urban sprawling and agricultural development is resulting in a swift ecosystem change in many parts of the country. Likewise, farmlands and human settlements are rapidly extending in the Ethiopian highlands where these areas are the source of many rivers and streams that discharge into the arid zone of the lowlands of Ethiopia (Jan et al., 2015). A result of human-induced problems such as the intensive expansion of urbanization and agricultural practices which were aggravated by rapid population growth resulted in a dramatic decline in the forest covers and severely threatened the upper head watersheds. The annual topsoil loss was estimated at 1.5 billion tons and over 82% of the total land area in Ethiopia is experiencing soil erosion (Ermias *et al.,* 2016).

The Legedadi and Dire watersheds are located approximately 35 kilometers northeast of Addis Abeba, in the Oromia regional state (Figure 1) covering 215 km^2 and 97.5 km^2, respectively. The Legedadi watershed is the largest of Addis Ababa's three main water supply sources. The average annual surface water potential of the two watersheds was estimated to be 86 million m^3 (MCM) for the Legedadi watershed and 50 MCM for the Dire watershed (AAWSA, 2011; 2016). The Legedadi and Dire watersheds are composed of various habitat types including forest, woodland, shrubland, grassland, and Afromontane. However, Eucalyptus plantation is the most dominating habitat in the area (Awraris, 2017).

The Dire and Legedadi areas are mainly gentle slopes covered with vegetation (trees and grass) down to the edges of the riverbanks where they locally become vertical rock escarpments just above the riverbed. The soil classification of the area is five types of soils in the watershed namely Vertic Cambisols, Orthic Solonaks, Chromic Vertisols, Eutric Nitosols, and Chromic Luvisols (AAWSA, 2011; 2016). Vertic Cambisols and Chromic Vertisols are the predominant soil types found in almost all parts of the watersheds (AAWSA, 2011; 2016).

The watersheds are characterized by the Moist Dega agro-climatic zone, receiving an annual rainfall range between 1,200 to 1,300 mm; the mean annual rainfall is 1233 mm. The rainfall is bimodal type, which is distributed into a minimum rainy season occurring in the months between March and April and a longer rainy season occurring between June and September. In Legedadi and Dire watersheds, the weather is relatively cool in the wet season of June to September, while October to May has warmer temperatures with easterly winds. The mean monthly temperature is between 14^{o}C and 18^{o}C throughout the year.

They are both sub-catchments of the Akaki River basin, which flows northeast to southwest and is part of the drainage system that forms the northwest corner of the Awash River basin.

The sub-catchment of Legedadi and Dire watersheds which mainly include the Lege Beri, Lege Sekoru, and Lege Bolo are from these sub-catchment's streams enter to Legedadi reservoir through a common course and some others merge several hundred meters before entering Legedadi reservoir. The Akaki River flows towards the southeast to join the larger Awash River (AAWSA, 2011; 2016). The altitude of the watersheds ranges between 2346 to 3245 MASL. The region is characterized by a range of volcanic mountains. The major physiographic units found in the watershed area are mountains, dissected side slopes of mountains, hills, steep to undulating foot-slopes, gullies, valleys, and undulating plains and flat to almost flat plains (AAWSA, 2011; 2016).

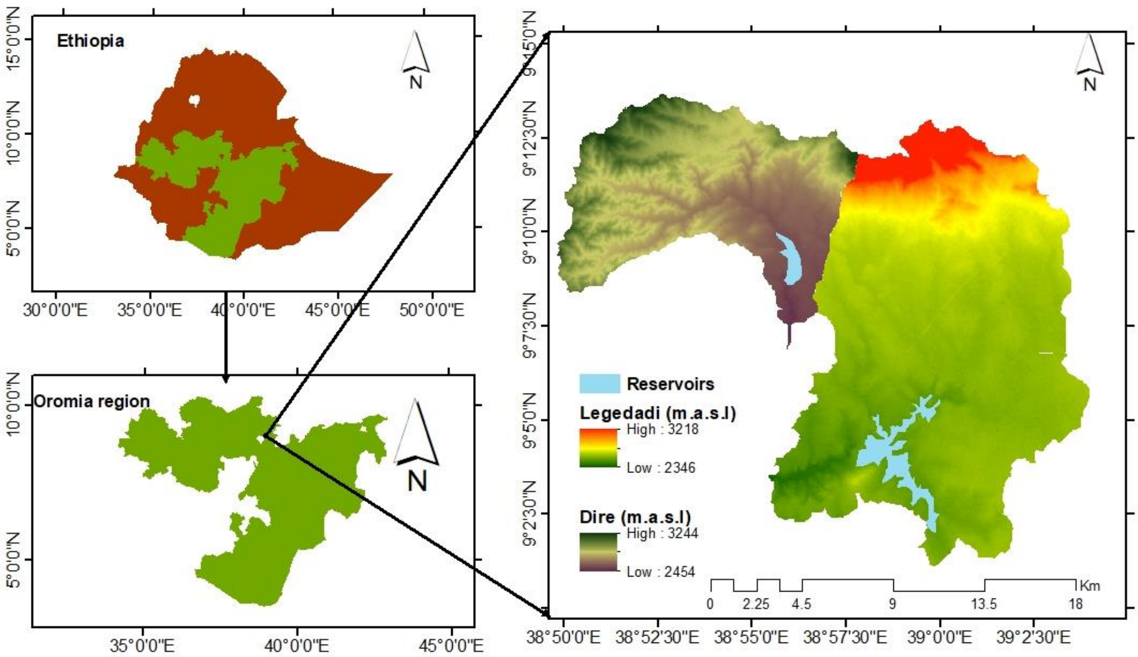

Figure 1: The Legedadi, and Dire watersheds and water reservoirs.

1.2 Literature Review

Initially, the concept of ecosystem services arose to illustrate the connection between human well-being and the natural environment (Ehrlich and Ehrlich, 1981). Later, the global value of ecosystem services was estimated (Costanza et al., 1997) to raise awareness about the economic value of ecosystems. The United Nations MEA, which was launched in 2001, paid more attention to the development of ecosystem services research to better understand the effects of ecosystem change (MEA, 2005). Since then, numerous studies on ecosystem services assessment, modeling, and valuation for conservation planning (Egoh et al., 2008; Boyd and Banzhaf, 2007), and land administration (Nelson et al., 2009) have been conducted.

Ecosystem services are generally categorized into four extensive classifications, namely provisioning, regulating, cultural, and supporting (De Groot et al., 2002; MEA, 2005) (Table 1). The provision of benefits to the ecosystem that directly impact humans is a significant element of the services that are provided. The advantages that are obtained from the management of ecosystem processes, which include climate, waste, and flooding, as well as the regulation of water and air quality, are commonly known as regulatory services. Non-material advantages that are derived from the ecosystem, such as spiritual inspiration, aesthetic experiences, education, and recreation, are also included. Supporting services, on the other hand, are the ecosystem functions that generate other services, such as primary production, nutrient cycling, and soil formation (MEA, 2005).

Table 1: Categories of Ecosystem services

Types of services	Description of products	Examples
Provisioning services	Ecosystem products and goods	• Water and grazing • Fuel wood and fiber
Regulating services	Regulatory services because of the ecosystem process	• Climate and flood regulation • Pollination
Cultural services	The non-material benefits people obtain from ecosystems	• Spiritual and religious • Recreation and ecotourism • Aesthetic, Inspirational, Educational
Supporting services	The basic services for the generation of all ecosystem services	• Soil formation • Nutrient cycling

	• Primary production • Habitat for endemic and rare species

Ecosystem services assessment is an important element in the application of ecosystem services in practice (Daily and Matson, 2008; Duarte et al., 2016). It also has been recognized as an important tool for communicating ecosystem conditions and services of various scales (Maes and Jacobs, 2017). Multiple ecosystem functions could occur in a larger landscape and seascape, providing enormous ecosystem services. Prioritization of spatial areas is also regarded as an important step in conservation planning (Knight et al., 2006). Land management decisions could incorporate areas with the optimal trade-offs between services, biodiversity, and economic activities (Maes et al., 2012). These are critical decision-making tools, particularly in conflict zones where economic activities have an impact on the natural environment (Duarte et al., 2016).

Most ecosystem service assessments solely focus on ecosystem service supply and lack an integrated consideration of the demand (Chang, 2018; Cheng et al., 2022). Such a disregard for ecosystem service demand creates a knowledge gap that has an impact on the quality of management and policy decisions. Understanding the ecosystem services produced, needed, and consumed, can be useful to minimize the gap between supply and demand (Troy and Wilson, 2006; Swetnam et al., 2011). The supply of ecosystem services in a specific area is the ability to provide ecosystem services (Burkhard *et al.,* 2012). On the other hand, ecosystem service demand reflects the services that are consumed or desired by users (Wolff et al., 2015).

Assessing the ecosystem service demand could help to understand the ecosystem quantity and the location where distinct ecosystem service is produced, needed, and consumed (Troy and Wilson, 2006; Swetnam *et al.,* 2011; Wolff *et al.,* 2015). On the other hand, quantifying and mapping the provision of ecosystem services in a particular area within a given period is critical (Burkhard *et al.,* 2012). More recently, given the importance of considering ecosystem services provision and demand several conceptual frameworks have been designed (Geijzendorffer *et al.,* 2015; Schröter *et al.,* 2014; Villamagna *et al.,* 2013).

Humans have altered the global ecology through the conversion of natural landscapes into other economic land used to support human populations (Grimm et al., 2008, Tylianakis et al., 2008, Kleijn et al., 2009; Ellis, 2015). Understanding the ecological impacts of land use change is critical for managing lands to maintain ecosystem integrity (Allen et al., 2019). Even though human land development can cause dramatic ecological changes, many aspects of an ecosystem remain in place (Li et al., 2007).

Long-term ecological research is essential for comprehending the temporal dynamics of land use change on ecosystems. LULC changes have a direct impact on biodiversity (Vitousek et al., 1997; Sala et al., 2000); and influence local and regional climate change (Houghton et al., 1999). Further, it is a primary cause of soil degradation (Tolba et al., 1992) and an increase in natural disasters such as flooding and drought (Mas et al., 2004; Dwivedi et al., 2005). Such changes also influence the vulnerability of people to climatic, economic, and sociopolitical disruptions (Kasperson et al., 1995). Ecosystems all over the world are constantly changing in structure, complexity, and type (Grimm et al., 2008, Tylianakis et al., 2008, Kleijn et al., 2009; Ellis, 2015). Changes in ecosystem composition result in ongoing changes to both floral and faunal communities, which can change underlying ecosystem processes such as seed dispersal, seedling establishment, and community structure.

The diversity observed among living organisms and the complex ecological systems comprising biodiversity have been the subject of much scientific inquiry and analysis (Mace *et al.*, 2012; Xu et al., 2019). The variability of species enhances ecosystem functionality and resilience which enhances ecosystem function and stability in ecological communities that significantly contributes to improving ecosystem goods (Oliver *et al.,* 2015). The relationship between biodiversity and ecosystem functioning has been influencing the provision of ecosystem services across scales (Cardinale *et al.,* 2012). However, both biodiversity and ecosystem functioning are influenced by the interactions between individuals or species (Harrison *et al.,* 2014; Liquete et al., 2016).

The state of biodiversity can be determined by habitat function (Basane and James, 2016). A high-quality habitat is considered a habitat that possesses high biodiversity and significantly contributes to the improved ecosystem processes and services (Johnson, 2007). A habitat is an area that contributes to the reproduction and the continuous existence of species (Sharp *et al.,* 2020). Habitat quality pertains to the capacity of an ecosystem to furnish favorable living circumstances that enable the survival and persistence of individual organisms and populations (Caro *et al.,* 2020; Liu et al., 2021). Habitat quality is one of the indicators of biodiversity in a particular area (Norliyana and Mohd, 2020). The quality of a given habitat is an essential component for ensuring the maintenance and existence of biodiversity. The ecosystem services are highly dependent on habitat quality; though, there may be differentiation within habitat quality (Thomas *et al.,* 2021).

Since biodiversity ensures ecosystem stability and ecological balance, an area endowed with abundant flora and fauna can provide higher ecosystem goods and services. The presence of high-quality habitats in each area is a positive indication that the areas possess greater biodiversity and have better potential to support the delivery of a suite of ecosystem services (Stolton *et al.,* 2010). A particular habitat with special ecological and biological values is essential to the functioning of the wider ecosystem processes. This area requires extraordinary protection to safeguard the special value, such as sustaining vital ecosystem processes.

The biodiversity state can be used as a basis proxy tool to measure the quality of an ecosystem (Havlicek and Mitchell, 2014; Hamere et al., 2021). Therefore, as a proxy for the condition of biodiversity, habitat quality is the ability of a given ecosystem to provide essential services and is a determinant for measuring ecosystem health (Polasky *et al.,* 2011; Villamagna *et al.,* 2013). The capability of an ecosystem depends on the intactness of the natural habitat and is significantly affected by the proximity to adverse land uses by humans and the intensity of the land uses (Fu *et al.,* 2013; Yan *et al.,* 2018). The decline in naturalness and semi-naturalness of habitat together with other unprecedented impacts on land cover changes including expansion of settlements, intensive farming practices, and artificial monoculture plantations have a significant negative impact on habitat quality (Zhang H. *et al.,* 2020; Hamere *et al.,* 2021). The long-term increase in land use change would affect the structure, complexity, and type of ecosystem composition resulting in the change in both floral and faunal communities that in turn alter essential ecosystem processes which leads to continuous decline in habitat quality (Su *et al.,* 2020).

Robust information on habitat quality is invaluable to making proper decisions on conservation planning (Rouget *et al.,* 2003; Baral *et al.,* 2014) including expansion of conservation intervention sites, and identification of principal habitat components that are vital for highest ecosystem services (Basane and James, 2016). Further determining the basis for ecosystem services such as the Service Providing Unit (SPU) (Luck *et al.,* 2003) and Ecosystem Service Providers (ESP) (Kremen, 2005) is essential to understand the broader scale interaction between the distribution of resources in a particular geographic scale, functional traits, and factors affecting ecosystem services such as land use changes.

However, the relationship between the quality of a given habitat for supporting various life forms and the associated ecosystem processes, goods, and services has been hampered by fundamental knowledge gaps and poorly studied (Ken, 2012). There is a lack of empirical evidence on the role of biodiversity in maintaining ecosystem services (Mertz *et al.,* 2007; Bastian, 2013). There is a dearth of research that has made use of empirical evidence to analyze the part played by biodiversity in furnishing ecosystem services (Mertz et al., 2007). Nonetheless, the precise quantitative associations between biodiversity, ecosystem constituents, mechanisms, and services remain insufficiently understood (Carpenter *et al.,* 2009; De Groot *et al.,* 2010; Ken, 2012). Furthermore, the facets of biodiversity encompassing functional, structural, and genetic components have been inadequately examined (Feld et al., 2009). The operation of the ecosystem remains an enigma regarding the function of each species in furnishing ecosystem services, as posited by De Groot et al. (2010) and Gonzalez-Redin et al. (2016). Biodiversity may arise because of ecosystem services (Boyd and Banzhaf, 2007). Conversely, biodiversity constitutes an essential ecosystem property and a necessary condition for various ecosystem services (Bastian, 2013). The comprehension of the relationships between biodiversity and ecosystem services has the potential to enhance the delivery of ecosystem services, while concurrently supporting biodiversity conservation, as highlighted by Maes et al (2012). A multitude of scholarly works have emphasized the pivotal role of biodiversity in maintaining healthy ecosystem functioning. Decreasing plant diversity can affect ecosystem functioning and is likely to reduce ecosystem services (Cardinale *et al.,* 2012; Loreau, 2010). Conversely, ecological restoration produces an increase in biodiversity and the provision of ecosystem services (Rey Benayas *et al.,* 2009). There exists a positive correlation between the diversity of species and certain facets of ecosystem operation, namely productivity, biomass, nutrient cycling, carbon flux, or nitrogen utilization, as indicated by studies conducted by Schwartz et al. (2000) and Isbell et al., (2011).

Biodiversity plays an important functional role in ecosystems (Hooper *et al.,* 2005; Benayas *et al.,* 2009; Norris, 2012). Researchers have been working to reveal the details of this role in experimental and observational ecosystem studies in which biodiversity and ecosystem function are relatively straightforward to quantify (Klein *et al.,* 2007; Tilman *et al.,* 2006). Nevertheless, there is still much research work required to examine the quantitative relationships between biodiversity and ecosystem functions (Raffaelli, 2006).

Habitat loss has consistent, adverse negative impacts on biodiversity, for example, negatively affects species richness and population abundance (Laurance *et al.,* 2002), genetic diversity (Aguilar *et al.,* 2008), pollination (Potts *et al.,* 2010), watershed management (Bruijnzeel, 2004), and carbon storage (Fargione *et al.,* 2008). Thus, determining the state of habitat and biodiversity, particularly species richness and abundance is required to make inferences on the association of species dispersal and habitat (Ceballos and Ehrlich, 2006).

The assessment of habitat quality serves as a significant parameter for the evaluation of regional ecological security, as elucidated by Liu et al. (2021). The degree of regional biodiversity and the provision of ecosystem services are factors that are directly linked to habitat quality, as expounded by Tang et al. (2020) and Zhu et al. (2020). Consequently, the study of habitat quality has gained widespread attention in the domain of ecological security research, as highlighted by Edmonds et al., (2021). Farley *et al.,* (2005) and Thomas and Nisbet (2007) reported that water flow regulation can be significantly improved with an increase in forest areas.

1.2.1 Conceptual Framework of the Research

The mapping, quantification, and modeling of ecosystem services are becoming an important component of research for the integration of ecosystem services into policy decision-making processes (Daily and Matson, 2008; Burkhard et al., 2012; Willemen et al., 2015; Adem et al., 2023). Ecosystem services mapping has been used to monitor conservation interventions (Egoh et al., 2008), and landscape planning (Albert et al., 2014; Polasky et al., 2011) over the last decade. maps and models can be used to visualize the spatial distribution of ecosystem services (Kareiva et al., 2011; Martnez-Harms and Balvanera, 2012; Crossman et al., 2013; Adem et al., 2023). The act of mapping the spatial extent and configuration of landscape features holds significant potential in identifying the type and value of ecosystem services available in a particular area (Jones et al., 2011; Verhagen et al., 2016; Sahle et al., 2018; Schmidt et al., 2019). Research on ecosystem services is expected to expand as the application of ecosystem services into the design of appropriate and efficient nature-based solutions programs (Maes and Jacobs, 2017).

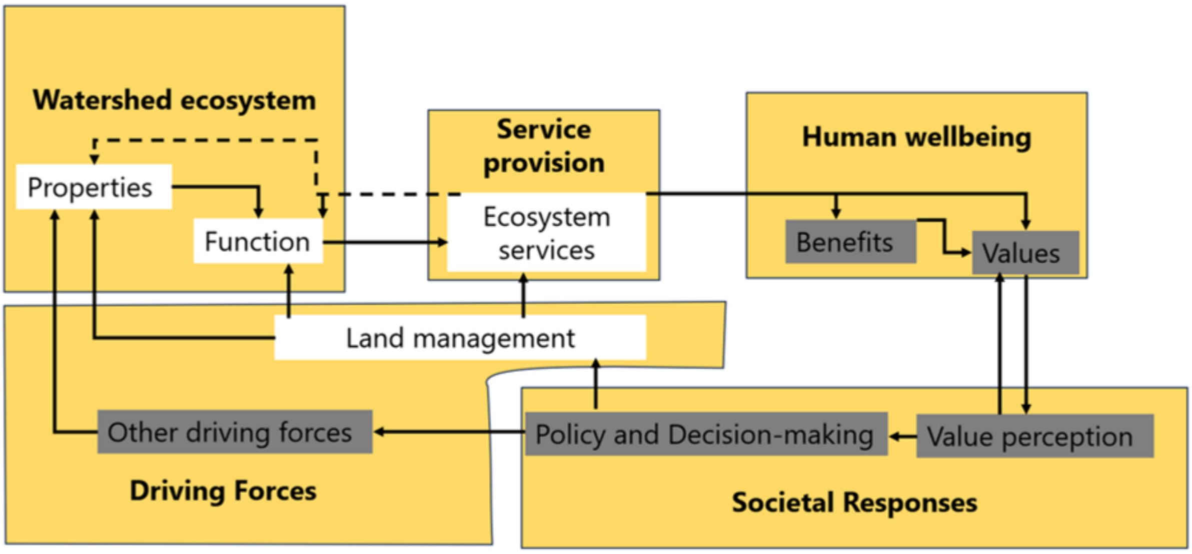

Figure 2: Conceptual framework of the study adopted from De Groot at al., 2010a; and Hein, 2010.

Following the conceptual framework presented (Figure 2), the study first assessed the impacts of LULC changes on Ecosystem Service Values (ESVs) in both watersheds. In the second paper, the quality of habitats was assessed based on the standardized modeling of InVEST. In the third paper, the study quantified the annual water yield capacity and then assessed the impacts of LULC changes and climate variability on the water yield capacity of the watersheds. The InVEST model was used to quantify and map water yield and the associated impacts exerted by climate and LULC changes on water yield were also assessed using the model. The annual water yield capacity of the watersheds was estimated based on the actual conditions.

In the fourth paper, the study assessed the recreational potential of the watersheds using the previously established recreational indicators namely, naturalness, greenness, level of disturbance, diverse landforms, activity, and water features (Nigussie *et al.,* 2021). Sustainable ecotourism planning and development interventions can be applied to improve the recreational services of the landscapes. In the fifth paper, land evaluation was made for apple-based agroforestry farming in both watersheds. Finally, the study provides premises for the establishment of market-based conservation mechanisms to sustainably finance sustainable watershed management interventions in the watersheds.

1.3 Statement of the Problem

Like most highlands of Ethiopia, the Dire and Legedadi watersheds have been threatened by several factors including habitat destruction and degradation due to the expansion of cultivation land, unrestricted grazing, and urbanization (AAWSA 2016; Awraris, 2017). The indigenous trees were dominantly covering the area but rapidly disappeared (Pohjonen, 1989). Natural forest native to the central highlands is under threat due to rapid population growth and urbanization (Legesse, 2010). Population growth, environmental problems, rapid industrialization, and the demand for natural resources are projected to increase (Abiye *et al.,* 2009). Thus, the water quality of reservoirs has declined (Taye, 2009; Yilikal et al., 2018; Helnata, 2019). Moreover, the radical outward settlement expansion to peri-urban areas is threatening the water sources (Tamiru *et al.*, 2005). The rapid population growth through natural birth and migration rates are expected to challenge the watershed's capacity to deliver drinking water for Addis Ababa city and the surrounding fast-growing towns (Tamiru *et al.*, 2005, AAWSA, 2011; 2016; Yilikal et al., 2019; Zinabu and Michael, 2020). The unmet water demand in Addis Ababa is projected to rise (Kifle *et al.*, 2017) to 307 million cubic meters in 2030 (Yilikal et al., 2019; Zinabu and Michael, 2020).

The population growth and subsequent development of marginal areas in Ethiopia further reduced the natural vegetation of the highland areas (Aster, 2004; Muleta *et al.*, 2006). This is severely threatening the production of agriculture and the long-term ecosystem sustainability in the watersheds. The unsustainable land use practices resulted in significant sediment load to the reservoirs (Gebreegziabher *et al.*, 2021). These alter the regular flow of water and pose a risk to potable water that affects the welfare of people and exposes the downstream people to social, economic, and environmental problems (Yilikal et al., 2018; Yilikal et al., 2019).

One of Africa's fastest-growing cities Addis Ababa requires a steady water supply to meet year-round residents and seasonal visitors. The city is a significant contributor to the country's GDP (Ezana, 2021). The chronically unbalanced water supply and demand cannot satisfy the current fast-growing population (Gebrekidan et al., 2023; Tirusew and Fekadu, 2023). Thus, it is very urgent to design effective watershed management practices in the headwaters. Sustainable management of the ecosystem intervention can be considered the most effective option to tackle the water scarcity in the city and its surroundings (Yilikal et al., 2019; Tirusew and Fekadu, 2023).

1.4 Research Questions

1. How do Land Use Land Cover changes affect the ecosystem service values across the years?
2. What is the status of habitat quality in terms of providing ecosystem services?
3. What is the annual water-yielding capacity of the watersheds?
4. How are Land Use Land Cover changes and Climate variability influencing the water yield in the watersheds across the years?
5. What is the extent of land available for outdoor recreational activities?
6. How much land is available for apple-based agroforestry farming to improve ecosystem services?

1.5 Objectives

1.5.1 General Objective

The prime objective of this study is to quantify and map ecosystem services to support the conservation and management of the Dire and Legedadi watersheds in the central highlands of Ethiopia.

1.5.2 Specific Objectives

1. Quantifying land use land cover changes and its impacts on ecosystem service values.
2. Quantifying and mapping habitat quality in responses to land use land cover dynamics.
3. Quantifying and mapping the annual water-yielding capacity of the watersheds.
4. Assessing the recreational potential of the watersheds.
5. Determining suitable land for apple-based agroforestry farming intervention.

1.6 Significance of the Study

Understanding the trajectories and extents of LULC is essential to develop invaluable information about the magnitude and trends of LULC changes for robust policy and decision-making related to land management and its proper implementation. The foremost significance of this study is that it sought to demonstrate the need for a management paradigm that comprised innovative market-based instruments to address overall biodiversity loss and sustainable natural resource management interventions. This study assessed the impact of land use changes on ecosystem services such as habitat quality, recreation, and water production, and further broader concepts have been covered including watershed management and local economic development. Further, the ecosystem service values assessment could potentially be invaluable to promoting the landscape's socioeconomic, political, and cultural importance and could contribute to finding additional financial and social support for the protection of landscapes. Finally, the study's findings will be critical in developing ecosystem services approaches to landscape restoration and sustainable livelihood schemes which could contribute to an improvement in providing a wide range of economic and environmental benefits.

1.7 Scope of the Study

This study was limited by both thematic and geographic scopes. Thus, the study assessed the impact of LULC changes on ESVs, HQ, and AWY; assessed the recreational potential and land suitable for apple farming, and finally assessed the premises of market-based conservation tools to use as management interventions in the most important watersheds of Dire and Legedadi watersheds in central Ethiopia.

1.8 Limitations of the Study

The quality and resolution of the data used can limit the assessment of LULC changes (Ashbir et al., 2020). Data were gathered from a variety of sources, which could be the study's main limitation because uncertainties in simulated LULC changes cannot be completely avoidable (Ashbir et al., 2020). Due to limited data sources, the impact of future land-use changes and other significant challenges, such as climate change, on the landscape ecosystem could not be assessed. An additional limitation of this study is the difficulty in valuing ecological changes using land cover types and the equivalent" biome category to which the assigned ESVs belong in a broader context, as well as the significant difference that can occur when applied in the local context. Further, the study applied the InVEST model, however, the model has various limitations including generating insufficient information about species distribution (Stephen et al., 2011). The InVEST model, albeit valuable, presents a notable limitation in that it does not incorporate the effects of soil, climate, hydropower generation, edge effect, and vegetation discrepancies on a given ecosystem (Zhang et al., 2014). Furthermore, ecological threat factor weighting depends on expert judgments (Terrado et al., 2016; Baral et al., 2014). Thus, expert's opinions had a significant impact on modeling habitat quality (Abreham et al., 2020).

1.9 Structure of the Dissertation

The thesis is organized into nine chapters, **Chapter one** provides background and justification of the research. A brief review of the literature was presented and statements of the problems, study significance, scope, and limitations were also presented. The s**econd chapter published in Heliyon** describes how the ESV changes in response to land cover dynamics; the analysis can help with the land management decision-making process. The **third chapter** published in the Journal of **Sustainable Environment** examines the impact of land cover changes on habitat quality. The **fourth chapter published in Water Conservation Science and Engineering** compares the impact of climate variability and land use changes on annual water yield. In **Chapter Fifth**, the ArcGIS tool was used to model the watershed's recreational potential using previously created indicators. **Chapter Six published**

in Heliyon examines land suitability for apple farming; the analysis aimed to support the management intervention for ecosystem service restoration and livelihood improvement. Finally, **Chapter Seven**, published in **Integrated Environmental Management and Assessment** synthesized the findings of the study.

Chapter Two

2 Impact of Land Use Land Cover Changes on Ecosystem Service Values in the Dire and Legedadi watersheds

2.1 Abstract

Land use land cover (LULC) change in a landscape is the main driver of degradation in ecosystem goods and services. This study was aimed at analyzing the dynamics of the LULC change in the catchments of the water supply reservoirs as well as the impact on the LULC Ecosystem Service Values (ESVs) between 1985 and 2022. The benefit transfer method was used to evaluate ecosystem service value (ESV) changes in response to LULC. The watersheds experienced substantial LULC changes. As a result, the natural vegetation, grasslands, and eucalyptus plantations declined dramatically, while settlements and cultivated lands increased substantially. The global and local ESV estimates show a dramatic decline in ESVs in the last three decades. According to global and local ESV estimates, total ESV in the Legedadi watershed has decreased from approximately US$ 65.8 million in 1985 to approximately US$ 11.9 million in 2022 and from approximately US$ 42.7 million in 1985 to approximately US$ 9.66 million in 2022. According to global and local ESV estimates, total ESV in the Dire watershed decreased from approximately US$ 437 thousand in 1985 to approximately US$ 59 thousand in 2022 and from approximately US$ 225 thousand in 1985 to approximately US$ 36 thousand in 2022. The overall decline in ESV demonstrates that the natural environment is deteriorating because of the replacement of the natural land cover by other economic land uses. Hence, it is highly recommended that implementing sustainable watershed management practices to halt the dramatic loss of natural ecosystems must be a high priority.

Keywords: Dire, ESV, Legedadi, LULC, Watersheds

2.2 Introduction

Humans have been continuously altering ecosystems in various ways. Most prominent is the conversion of natural areas into farms and urban areas, which is the primary cause of ecosystem degradation in many places (Grimm et al., 2008; Tylianakis et al., 2008; Kleijn et al., 2009; Eyasu et al., 2019). These changes have a negative impact on climate patterns, biodiversity resources, and socioeconomic dynamics (Sala et al., 2000), resulting in biodiversity loss, global warming, and increased natural disasters such as flooding and drought (Sala et al., 2000; Mas et al., 2004; Dwivedi et al., 2005). Changes in land use and land cover (LULC) have negative impacts on ecosystem services (Bennett and Saunders, 2010). However, ecosystem functions can be improved by biodiversity resources that support the stability of ecological communities (Stolton *et al.*, 2010; Oliver *et al.,* 2015).

The existence of reliable information on LULC changes is essential for land management and planning (Fan et al., 2007). The rapidly growing population and increased socioeconomic demand have put pressure on LULC not only in unprotected lands but also in nature reserves, protected habitats, and national parks (Amsalu et al., 2007; Efrem et al., 2009; Geremew, 2013; Mesfin and Kumelachew, 2018; Belay and Mengistu, 2019). For instance, in Ethiopia, significant LULC changes have been observed (Gete and Hurin, 2001; Belay, 2002; Girmay, 2003; Selamyihun, 2004; Emiru et al., 2019) in various landscapes under legal protection such as the Bale Mountain Eco-Region (Sisay et al., 2016), the Awash National Park (Solomon et al., 2014), Simien Mountains National Park (Menale et al., 2011), and Nech Sar National Park (Aramde et al., 2016). These modifications are linked to deforestation, biodiversity loss, and land degradation (Selamyihun, 2004).

Understanding landscape patterns and the effects of land conversion on natural phenomena is critical for the proper land management decision-making process (Rawat and Kumar, 2015).

The availability of reliable information on the values of natural ecosystem services can make the contribution of natural areas more visible and quantifiable at the highest levels of decision-making (Hailu et al., 2020). More recently, several studies have estimated the ecosystem service values (ESVs) of important landscapes by assigning ESV coefficients to different LULC types as a proxy (Costanza et al., 1997; van der Ploeg and de Groot, 2010; Knoke et al., 2011; Kindu et al., 2016; Terefe et al., 2016; Terefe et al., 2017; Hailu et al., 2019; Abreham et al., 2019; Ashbir et al., 2020; Hailu et al., 2021; Wondimagegn et al., 2022). Valuing ecosystem services in monetary terms is critical for increasing user awareness and presenting evidence to decision-makers (Arowolo et al., 2018; Alarcon et al., 2016). This would assist decision-makers in sustainably managing environmental resources (Biedemariam et al., 2022). The Dire and Legedadi watersheds have considerably changed in LULC in which natural forests and grasslands were replaced by farmland and settlements (AAWSA, 2016; Awraris, 2017). Thus, assessing the effects of LULC changes on ESVs in the Dire and Legedadi watersheds can help increase public awareness (Temesgen et al., 2018). Further, the information can be used to support the decision and policymaking in landscape planning and management of ecosystem services (de Groot *et al.*, 2010; Temesgen et al., 2018) and to predict the implications of land use management on the ecosystem services (Lawler *et al.*, 2014). The objective of this study was, therefore, to quantify LULC changes and assess the economic value of ecosystem service changes associated with LULC dynamics in the most important watersheds of Addis Ababa city by using time series Landsat images for the period between 1985 and 2022.

2.3 Materials and Methods

2.3.1 Data collection

Geographic Information Systems (GIS) and Remote Sensing (RS) techniques are powerful and low-cost tools for assessing spatial and temporal changes in LULC (Herold et al., 2003;

Serra et al., 2008). The application of remote sensing data is the most common method for detecting, quantifying, and mapping LULC patterns in a landscape (Chen et al., 2005). This study employed Landsat series images 5, 7, and 8 Operational Land Imager (OLI) and Thermal Infrared Sensor (TIRS) images with 30 m spatial resolution to detect the LULC changes between the years 1985 and 2022. The analysis focused on this period because there have been unregulated LULC changes attributed to the transformation of government policies associated with land-related resources in the assessment period (Tatek, 2022) and to be able to illustrate the trends over the last four decades.

2.3.2 Image processing

The Landsat images of the years 1985, 1995, 2010, and 2022 were freely obtained from the United States Geological Survey (USGS) website (URL: https://www.usgs.gov/landsat-missions/landsat-data-access). To limit the impact of the wet season variation in vegetation and to minimize cloud cover, all the acquired images were taken for the dry season (January-February). Before classification, the images were converted to the RGB color composite to get a better view of surface features and to clearly distinguish them during classification (Asmamaw, 2013; Kazakeviciute-Januskeviciene *et al.*, 2020). All the images were projected to the Universal Transverse Mercator (UTM) zone 37 N and datum of World Geodetic System 84 (WGS84) for consistency and compatibility.

2.3.3 LULC classification

The accuracy of the classified images was checked against reference data collected from the field sites and other images (Foody, 2002). During the field survey, Ground Control Points (GCPs) were collected from representative locations and through participant observation methods by traversing the landscape through transect walks with local communities (Muzein, 2006). By using the collected reference points, a comprehensive accuracy assessment was conducted, which included an examination of overall accuracy, user accuracy, and producer accuracy (Table 2). The total accuracy was calculated by dividing the number of correctly classified elements (i.e., the sum of the diagonal elements in an error matrix) by the total number of pixels included in the evaluation process (Foody, 2002; Lillesand et al., 2015; Aramde et al., 2016). The Kappa statistic is an alternative measure of classification accuracy that subtracts the effect of random accuracy and quantifies the magnitude of correct classification compared to another (Foody, 2002; Lillesand et al., 2015; Aramde et al., 2016).

Table 2: Accuracy assessment

Land class	1985		1995		2010		2022	
	Producer	**User**	**Producer**	**User**	**Producer**	**User**	**Producer**	**User**
Natural vegetation	76.50	59.10	78.95	78.95	88.24	78.95	88.24	78.95
Settlement	86.20	80.60	87.10	90.00	89.74	92.11	91.84	93.75
Grassland	100.00	70.00	100.00	82.61	100.00	82.61	100.00	82.61
Eucalyptus Plantation	50.00	68.80	78.95	88.24	82.61	90.48	85.19	92.00
Cultivation	56.80	51.20	72.41	63.64	77.78	73.68	81.40	77.78
Waterbody	62.50	100.00	76.92	100.00	76.92	100.00	76.92	100.00
Bare land	70.60	70.60	85.71	85.71	85.71	100.00	85.71	100.00
Overall, accuracy	67.50	67.50	81.51	81.51	85.71	85.71	87.36	87.36
Kappa statistics	0.60	0.60	0.78	0.78	0.83	0.83	0.85	0.85

2.3.4 LULC change detection.

Change detection is a common method for analysing LULC changes (Hegazy and Kaloop, 2015). Image overlay, classification comparisons of land cover statistics, change vector analysis, principal component analysis, image rationing, and normalized difference vegetation index (NDVI) differencing are the most used change detection methods (Jensen, 2000). The following formula (Eq.1) was used to compute the magnitude of the LULC change (Fasika et al., 2019). Each type of LULC in the study period was compared and the rate of change was determined using the equation (Eq.2).

$$\text{Total area change} = \text{Area}_{\text{final year}} - \text{Area}_{\text{initial year}} \ldots\ldots\ldots\ldots 1$$

Where an area is the extent of each LULC type; positive values suggest an increase whereas negative values indicate a decrease (Hamere et al., 2021).

$$\text{Rate of change } = \frac{\text{Area final year} - \text{Area initial year}}{N} \ldots\ldots\ldots\ldots\ldots\ldots 2$$

Where N is the time interval between the initial and final years.

2.4 Global and local ESV coefficients

The total ESVs in the watersheds during the study period were calculated using two types of ESV coefficients of the respective LULC types. The ESVs coefficients were used as a proxy for each LULC type in the global coefficients (Costanza et al.,1997). Local-level estimates of ESV are crucial in Ethiopian landscapes because of the diversity of ecosystem services resulting from heterogeneity in land use characteristics (Getachew et al., 2014; Kindu et al., 2016). Thus, the total ESVs in both watersheds were estimated using locally modified coefficients adapted from an earlier study (Kindu et al., 2016) (Table 3).

2.5 ESV analysis

The benefits transfer technique, which is based on worldwide value coefficients or adjusted value coefficients has been a widely used method, especially in data-deficient areas (Costanza et al., 1997; Kindu et al., 2016). The ESVs analysis was performed using coefficients assigned to each watershed's LULC type (Table 3). The total ESV for each LULC type was calculated by multiplying the area (ha) of each LULC type by its corresponding value coefficients (Costanza et al., 1997; Kindu et al., 2016) using the formula below in Eq. (3). To analyse the impact of variations of ESVs over time, the obtained ESV was also adjusted to the single year 2017 using the US Bureau of Labour Statistics CPI Inflation Calculator.

$$ESVt = \sum(Ak * VCk)\ldots\ldots\ldots\ldots\ldots\ldots 3$$

Where ESVt = Total estimated ESVs, Ak = area (ha) and VCk = the value coefficient (approx..... US$ ha^{-1}year^{-1}) for LULC type 'k'(Costanza et al., 1997; Kindu et al., 2016).

The ESV change was calculated using the formula below shown in Eq. (4) (Cabral *et al.*, 2016; Kindu *et al.*, 2016; Gashaw *et al.*, 2018).

$$ESV\ change\ (\%) = \frac{(ESVrecent\ year - ESV\ initia\ year)}{ESVinitial\ year} * 100\ldots\ldots\ldots\ldots 4$$

Although each LULC type did not perfectly match the biome proposed by Kreuter et al., (2001), the matching of the TEEB proposed biome and the LULC type in the watershed qualified this study. The tropical forest, for example, is assigned to natural vegetation, while the Eucalyptus plantation is assigned to woodland; settlement is assigned to the built-up area; farmland is assigned to cultivated land; reservoirs are assigned to a water body; and desert is assigned to bare land, while grass/rangeland is assigned to grazing land (Abreham et al., 2019). As a result, the individual ecosystem valuation was created using data obtained from Van der Ploeg et al., (2010) and cascaded into the East Africa individual coefficient of ESV function (McVittie and Hussain, 2013) (Table 4).

Table 3: ESVs coefficients (a) global ESV coefficients and (b) local ESV coefficients

Land classes	Equivalent biome	US$ ha^{-1} yr^{-1}	
		a	b
Bare lands	Desert	0	0
Natural forests	Tropical forest	2008	986.69
Plantation forests	Tropical forest	2008	986.69
Croplands	Cropland	92	225.56
Grasslands	Grass/rangelands	244	293.25
Settlements	Urban	0	0
Water	Lakes/rivers	8498	8103.5

Table 4: Ecosystem functions value coefficients of each LULC type.

Ecosystem function types	Ecosystem service values (US$ $ha^{-1}year^{-1}$ in each land use land cover							
	FL	CL	GL	WL	SL	BL	S/B	Wb
Water supply	178	0	114.4	4.33	114.4	0	0	1770.6
Food production	67	186	130	0	130	0	0	14.5
Flaw material	475.51	103.4	19	138.2	19	0	0	1.4
Biomass fuel	31.34	0	79	67.28	67.28	0	0	0
Genetic resources	327.68	0	0.01	0	0.01	0	0	8.23
Water regulation	15.6	0	5	0	5	0	0	3878.4

Water purification	275.2	0	99.71	253.56	99.71	0	0	1348.13
Erosion control	360	3.45	55.89	26.76		0	0	0
Biological control	20.62	30	30	0	30	0	0	0
Biodiversity protection	263	24	52.68	11.44	52.68	0	0	361.4
Climate regulation	569	204.65	121.91	383.72	121.91	0	0	66.76
Gas control	22.12	0	2.41	0	2.41	0	0	0
Carbon sequestration	1229.79	96	297.39	8.16	297.39	0	0	00 .00
Nutrient recycling	11.61	0	0	0	0	0	0	0
Pollination	48.29	20	32	0	0	0	0	0
Soil formation	3.26	0	7	0	0	0	0	0
Habitat/regulation	28.057	0	0	0	0	0	0	0
Recreation	810.68	0	3.11	0	0	0	0	519.47
Tourism	73.74	0	0	0	0	0	0	0

Note: *FL= Forest; CL=Cultivation; GL= Grassland; WL= Woodland; SL= Shrub/Bushland; BL= Bareland; S/B= Settlement/ Built area; Wb= Water body*

2.6 Elasticity of the ESV changes with LULC.

The elasticity of ESV change was calculated to assess the dependability of the ESV results. Using the value coefficient constant for the remaining LULC types identified, the ESV used in Eqs. (3) of a specific LULC class was adjusted by 50% (Mattias et al., 2017). Changes in the Coefficient of Sensitivity (CS) value should be small and < 1(Aschonitis et al., 2016). A simplified method developed by Aschonitis et al., (2016) was used to calculate the CS as shown in Eq. (5),

$$CS = \frac{VC_{ik}A_k}{ESV_i} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots 5$$

Where ESVi = the total ESV (approx. US$ yr^{-1}) from all LULC types identified at i year (approx. US$ yr^{-1}), VCi,k = total value of ES provided by the k LULC class at i year (approx.. US$ yr^{-1}), and Ak= area coverage (ha) of the k LULC type at i year.

2.7 Results

2.7.1 LULC changes (1985-2022)

During the study period, substantial LULC change was observed in both watersheds. Between 1985 and 2022, the natural vegetation, water body (Dire reservoir), and eucalyptus plantation in the Dire watershed have declined by 89.5%, 19%, and 28.5%, with annual decline rates of 1.15 km^2, 0.002 km^2, and 0.21 km^2, respectively. Whereas settlement, cultivation, and bare land have increased by 3025%, 136.6%, and 180%, respectively, with annual increment rates of 0.32 km^2, 0.33 km^2, and 1.99 km^2. Between 1985 and 1995, the Dire watershed's grassland-dominated land cover was largely replaced by bare land between 2010 and 2022. (Figure 3; Table 5). Similarly, the natural vegetation, grassland, and eucalyptus plantation declined dramatically, while settlement and cultivation increased drastically. The natural vegetation, water body (Legedadi reservoir), grassland, and eucalyptus plantation in the Legedadi watershed have declined by 53.8%, 26.42%, 97.77%, and 52.22%, with annual decline rates of 0.6 km^2, 0.02 km^2, 3 km^2, and 0.62 km^2, respectively. Whereas the cultivated land and settlements increased at a rate of 0.65 km^2 and 3.63 km^2 per year, respectively (Figure 4; Table 5).

The land transition matrix in both watersheds reveals that the conversion of natural vegetation to cultivation, settlement, and bare land has taken place. Similarly, the grassland has transformed into cultivation, settlement of Eucalyptus, and bare land. It is important to note that the land use types of settlement, cultivation, and water bodies have remained unchanged, except in the case of cultivation in the Dire watershed, which has been converted into bare land (Tables 6 and 7).

Table 5: LULC changes in the Legedadi and Dire watersheds (1985, 2022)

LULC types	**Legedadi watershed**							
	1985		**1995**		**2010**		**2022**	
	Area (km^2)	**%**	**Area (km^2)**	**%**	**Area (km^2)**	**%**	**Area (km^2)**	**%**
Natural vegetation	41.43	19.2	34.3	15.89	33.87	15.73	19.13	8.86
Waterbody	3.86	1.79	3.67	1.7	3.64	1.69	2.84	1.32
Settlement	0.2	0.09	1.98	0.92	23	10.68	24.38	11.29
Grassland	117.26	54.33	78.73	36.48	16.12	7.48	4.95	2.29
Eucalyptus	44.27	20.51	46.93	21.74	10.24	4.75	21.15	9.8
Cultivation	8.81	4.08	50.22	23.27	128.5	59.66	143.4	66.44
Total	**215**	**100**	**215**	**100**	**215**	**100**	**215**	**100**
	Dire watershed							
Natural vegetation	47.75	49.16	22.50	23.19	13.66	14.07	5.00	5.15
Settlement	0.4	0.41	1.37	1.41	6.79	6.99	12.5	12.87
Grassland	11.83	12.18	47	48.44	0	0	0	0
Bare land	0	0	0	0	13.25	13.64	37.14	38.25
Eucalyptus	28	28.83	16	16.49	26.95	27.75	20	20.6
Cultivation	9.15	9.42	10.15	10.46	35.63	36.69	21.65	22.3
Waterbody	0	0	0	0	0.84	0.86	0.81	0.83
Total	**97**	**100**	**97**	**100**	**97**	**100**	**97**	**100**

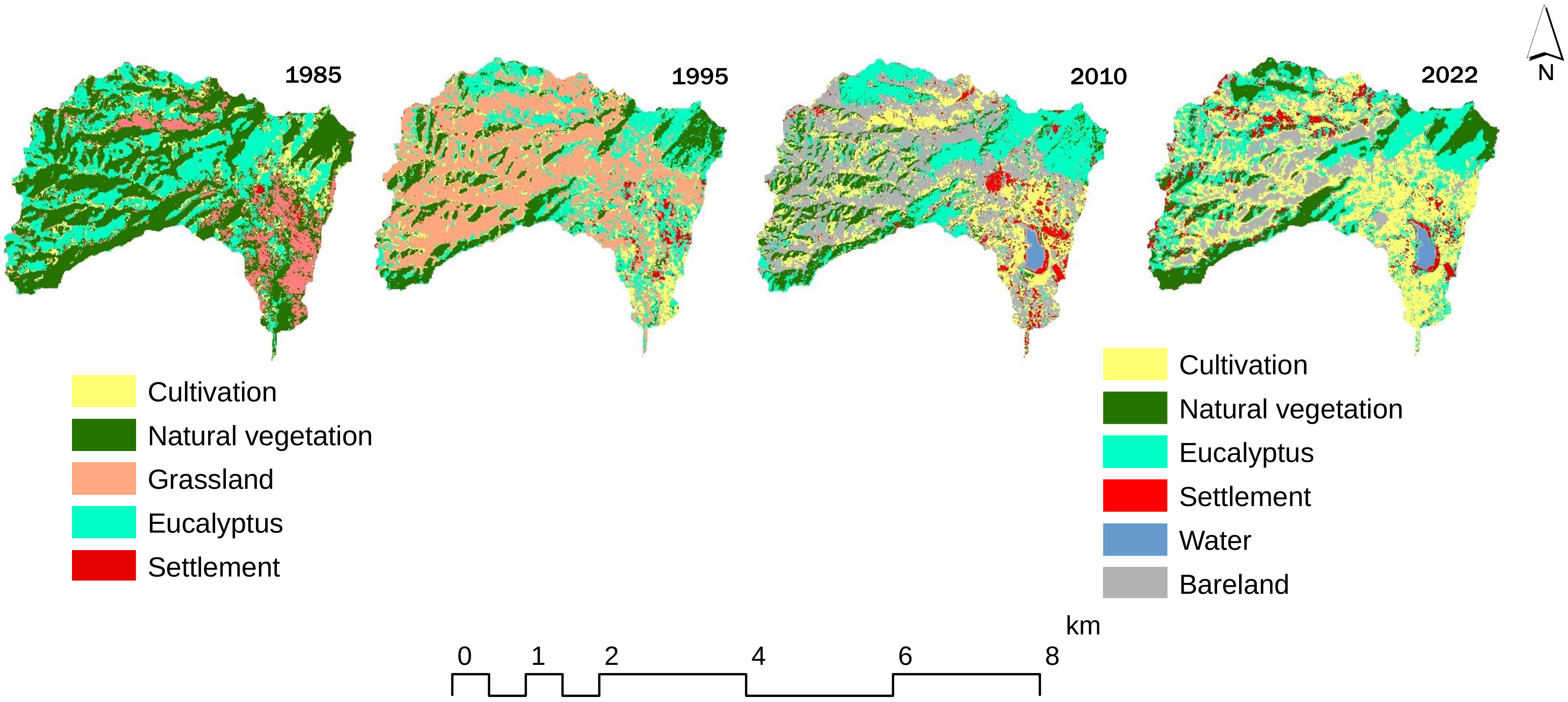

Figure 3: LULC changes in the Dire watershed (1985-2022)

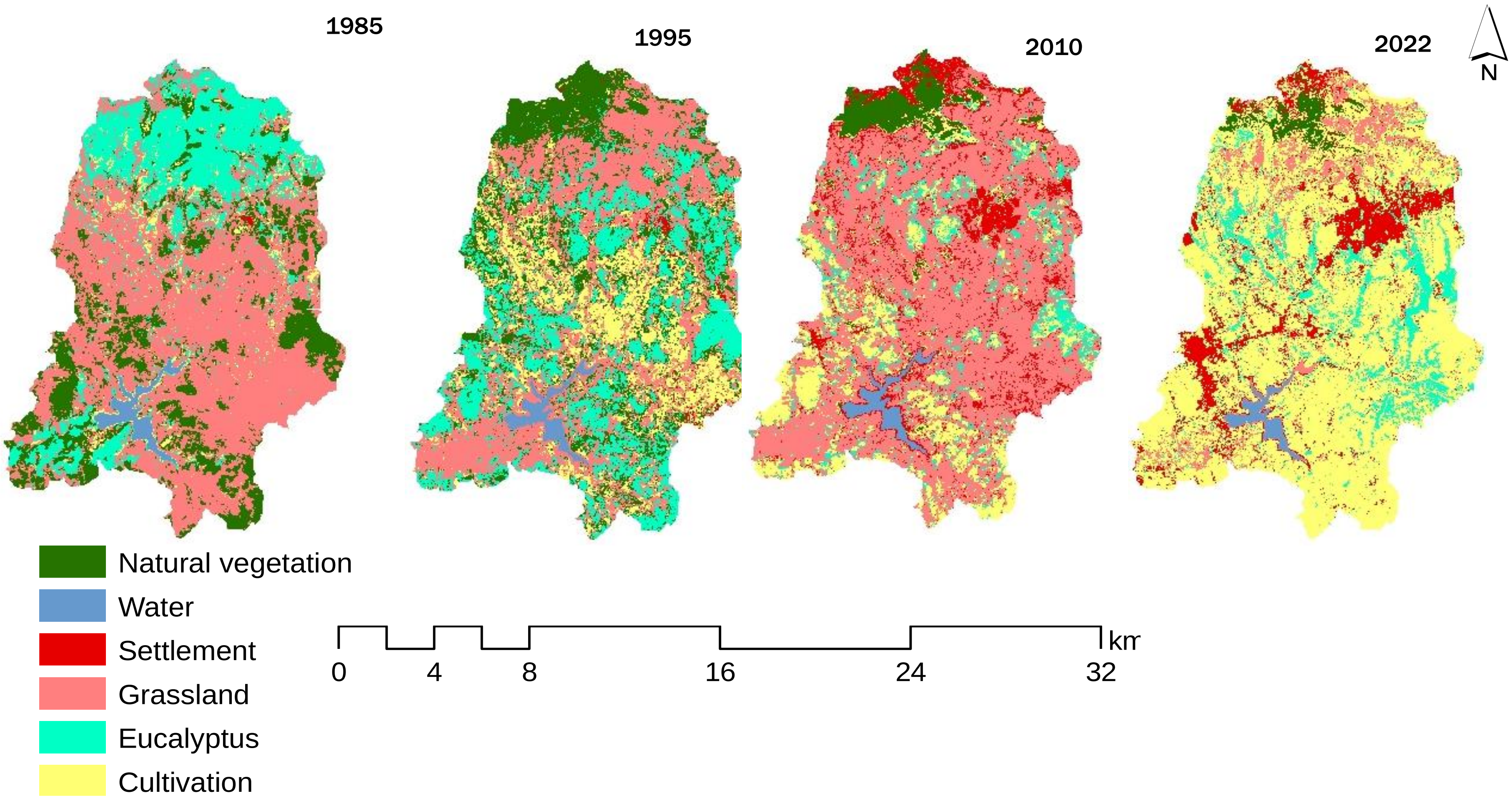

Figure 4: LULC changes in the Legedadi watershed (1985-2022)

Table 6 : Transition matrix of LUCC in the Dire watershed between 1985 and 2022 (unit: km^2)

	Land class	Land class 2022						
		Natural vegetation	Settlement	Bareland	Eucalyptus	Cultivation	Water body	Total
Land class 1985	Natural vegetation	**11.06**	2.32	14.09	11.42	8.19	0.44	47.52
	Settlement	0.01	**0.08**	0.09	0.03	0.23	0.00	0.43
	Grassland	0.38	1.50	**2.59**	0.87	6.51	0.13	11.97
	Eucalyptus	1.01	0.97	14.18	**7.27**	4.15	0.19	27.79
	Cultivation	0.13	0.29	5.70	2.16	**1.12**	0.05	9.45
	Total	12.59	5.15	36.65	21.76	20.2	0.81	97.17

Table 7: Transition matrix of LUCC in the Legedadi watershed between 1985 and 2022 (unit: km^2)

	Land class	Land class 2022						
		Natural vegetation	Waterbody	Settlement	Grassland	Eucalyptus	Cultivation	Total
Land class 1985	Natural vegetation	**1.31**	0.00	5.01	1.06	0.86	33.48	41.72
	Waterbody	0.00	**2.82**	0.13	0.80	0.00	0.09	3.86
	Settlement	0.00	0.00	**0.18**	0.00	0.00	0.03	0.21
	Grassland	0.50	0.00	12.71	**4.95**	17.85	80.15	116.16
	Eucalyptus	2.88	0.00	5.35	11.62	**2.29**	22.40	44.54
	Cultivation	0.27	0.00	1.49	1.73	1.32	**4.60**	9.41
	Total	4.96	2.82	24.88	20.17	22.32	140.75	215.90

2.7.2 Estimated ESVs (1985-2022)

Between 1985 and 2022, both global and local estimates of ESV coefficients show a dramatic decrease in ESVs. Based on the global and local ESV estimates, total ESVs in the Legedadi watershed have decreased from approximately US$ 65.8 million in 1985 to approximately US$ 11.9 million in 2022 and from approximately US$ 42.7 million in 1985 to approximately US$ 9.66 million in 2022 (Table 8). Similarly, based on global and local ESV estimates, total ESVs in the Dire watershed decreased from approximately US$ 437 thousand in 1985 to approximately US$ 59 thousand in 2022 and from approximately US$ 225 thousand in 1985 to approximately US$ 36 thousand in 2022 (Table 9).

Based on global estimates, the eucalyptus plantation was the leading contributor to the ESV in the Legedadi watershed in 1985, 1995, and 2022, accounting for approximately US$ 24.9 million (38%), US$ 18.5 million (43.2%), and US$ 4.2 million (35.5%), respectively, followed by natural vegetation accounting for approximately US$ 23.3 million (35.5%), US$ 13.5 million (31.58%), and US$ 3.84 million (32.17%). However, in 2010, natural vegetation contributed approximately US$ 9.3 million (50.3%), followed by the Legedadi water reservoir, which contributed approximately US$ 4.2 million (22.87%) (Table 8). Between 1985 and 1995, the eucalyptus plantation contributed approximately US$ 12.2 million (28.7%) and approximately US$ 9.1 million (32.1%), respectively, while natural vegetation contributed approximately US$ 11.4 million (26.86%) and approximately US$ 6.6 million (23.45%) (Table 8). Natural vegetation and cultivation, on the other hand, were the highest contributors, contributing approximately US$ 4.5 million (31.31%) in 2010 and approximately US$ 3.2 million (33.5%) in 2022, respectively, followed by the Legedadi water reservoir, which contributed approximately US$ 4 million (27.64%) and US$ 2.3 million (23.84%) in 2010 and 2022, respectively (Table 8).

Similarly, natural vegetation was the primary contributor to the ESV in the Dire watershed between 1985 and 1995, accounting for approximately US$ 269 thousand (61%) and US$ 89 thousand (58.69%) based on the global estimate, and approximately US$ 132 thousand (50.36%) and US$ 43 thousand (41.07%) based on the local assessment, respectively; eucalyptus plantation contributed approximately US$ 157.9 thousand (36%) and US$ 63 thousand (35%) (Table 9). However, between 2010 and 2022, the eucalyptus plantation became the main contributor to the ESV in the Dire watershed, accounting for approximately US$ 74 thousand (58.85%) and US$ 40 thousand (68%); and approximately US$ 36.4 thousand (48.42%) and US$ 19.7 thousand (54.64%) based on global and local estimates, respectively, followed by natural vegetation and water reservoir contributing approximately US$10 thousand (17%) and US$ 6.5 thousand (18.8%) in 2022 (Table 9).

Table 8: Estimates of ESVs of Legedadi watershed in million approx. US$ for each LULC type (1985-2022). ESV [a] = ESVs (million in 2022 approx.US$ yr $^{-1}$)

LULC types	1985		1995		2010		2022	
	ESV[a] (**%**)	ESV[b] (**%**)	ESV[a] (**%**)	ESV[b] (**%**)	ESV[a] (**%**)	ESV[b] (**%**)	ESV[a] (**%**)	ESV[b] (**%**)
Natural. Ve	23,376,794 (35.5)	11,486,877 (26.86)	13,568,256 (31.58)	6,667,163 (23.45)	9,317,501 (50.28)	4,578,429 (31.31)	3,841,304 (32.17)	1,887,537(19.55)
Eucalyptus	24,979,258 (37.94)	12,274,295 (28.7)	18,564,381 (43.2)	9,122,156 (32.1)	2,816,983 (15.2)	1,384,207 (9.47)	4,246,920 (35.56)	2,086,849 (21.61)
Cultivation	227,756 (0.35)	558,398 (1.31)	910,187 (2.12)	2,231,541 (7.85)	1,619,614 (8.74)	3,970,871 (27.16)	1,319,280 (11.05)	3,234,530 (333.5)
Grassland	8,039,814 (12.21)	9,662,605 (22.6)	3,784,393 (8.81)	4,548,251 (16)	538,859 (2.91)	647,625 (4.43)	120,780 (1)	145,158 (1.5)
Settlement	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Waterbody	9,217,440 (14)	8,789,542 (20.55)	6,143,969 (14.3)	5,858,749 (20.61)	4,237,782 (22.9)	4,041,053 (27.64)	2,413,432 (20.21)	2,301,394 (23.85)
Total	**65,841,065.04**	**42,771,718.57**	**42,971,188.42**	**28,427,862.27**	**18,530,740.56**	**14,622,186.17**	**11,941,716.00**	**9,655,470.47**

Table 9: Estimates of ESVs of Dire watershed in thousand approx. US$ for each LULC type (1985-2022). ESV [a] = ESVs (million in 2022 approx. US$ yr. $^{-1}$)

LULC types	1985		1995		2010		2022	
	ESV[a] (**%**)	ESV[b] (**%**)	ESV[a] (**%**)	ESV[b] (**%**)	ESV[a] (**%**)	ESV[b] (**%**)	ESV[a] (**%**)	ESV[b] (**%**)
Natural	269,428 (61.53)	132,391 (58.7)	89,004 (50.36)	43,735 (41.07)	37,578 (29.83)	18,465 (24.54)	10,040 (17)	4,933 (13.66)
Eucalyptus	157,989 (36.1)	77,632 (34.42)	63,292 (35.81)	31,100 (29.2)	74,138 (58.85)	36,430 (48.42)	40,160 (68)	19,733 (54.64)
Cultivation	2,365 (0.55)	5,799 (2.57)	1,839 (1.04)	4,510 (4.24)	4,490 (4.56)	11,010 (14.64)	1,991 (3.37)	4,883 (13.52)
Grassland/Bare	8,111 (1.85)	9,748 (4.32)	22,591 (12.78)	27,152 (25.5)	0 (0)	0 (0)	0 (0)	0 (0)
Settlement	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Waterbody	0 (0)	0 (0)	0 (0)	0 (0)	9,779 (7.76)	9,325 (12.4)	6,883(11.6)	6,563 (18.18)
Total	**437,894 (100)**	**225,572 (100)**	**176,728 (100)**	**106,497 (100)**	**125,986 (100)**	**75,230 (100)**	**59,075 (100)**	**36,114 (100)**

2.7.3 Impacts of LULC Changes on Distinct Ecosystem Service Values

Table 10 below shows the ESVs provided by different ecosystem services. The data shows that LULC changes have varying effects on the various ecosystem services in the study area. The total provisioning of ecosystem services in the Legedadi and Dire watersheds has decreased from US$ 103.6 thousand in 1985 to US$ 73.4 thousand in 2022; and from US$ 64.1 thousand in 1985 to US$ 17.3 thousand in 2022, respectively. In the Legedadi, regulating services have decreased from US$ 247.4 thousand in 1985 to US$ 88.5 thousand in 2022, while in the Dire watershed, it has decreased from US$ 161.8 thousand in 1985 to US$ 39.8 thousand in 2022. In the Legedadi and Dire watersheds, supporting services have decreased from US$ 8.5 thousand in 1985 to US$ 3.06 thousand in 2022, and from US$ 5,000 in 1985 to only US$ 889 in 2022, respectively. Cultural services have decreased from US$ 39 thousand in 1985 to US$ 1.5 thousand in 2022, and from US$ 42.2 thousand in 1985 to US$ 4.8 in 2022, in the Legedadi and Dire watersheds, respectively. Ecosystem services have declined across study periods, except food production, which increased from US$ 19.7 thousand in 1985 to US$ 28.6 thousand in 2022 in the Legedadi and increased from US$ 6.4 thousand in 1985 to US$ 9.5 in 1995 and to US$ 7.5 in 2010 before declining to US$ 4.3 in 2022.

Between 1985 and 1995, the value of regulating services accounted for more than 60% of the total in the Legedadi watershed, while it slightly decreased between 2010 and 2022. Similarly, between 1985 and 2022, the Dire regulating services accounted for the largest share, accounting for more than 59%. In all periods, the provisioning services are the second largest in both the Legedadi and Dire watersheds. It accounted for approximately 27% of the Legedadi between 1985 and 2010, rising to 44% by 2022. Similarly, provisioning services accounted for approximately 26% of all periods in the Dire watershed.

Cultural and support services have contributed to the lowest levels in both watersheds. Cultural and supporting services in the Legedadi contributed about 10% and 2% all the time, respectively, with cultural services dropping to 1% in 2022. Throughout the study period, cultural and supporting services contributed more than 11% and nearly 2% to the Dire watershed.

Table 10: Ecosystem functions value coefficients of each LULC type in the Legedadi and Dire watersheds.

Group	Indicators	Legedadi				Dire			
		Ecosystem service values (US$ year)				Ecosystem service values (US$ year)			
Provisioning services		ESV 1985	ESV 1995	ESV 2010	ESV 2022	ESV1985	ESV1995	ESV2010	ESV2022
	Water supply	27,815.29	21,813.42	14,362.31	9,091.50	9,974.09	9,451.08	4,035.48	2,410.79
	Food production	19,714.24	21,927.14	28,318.67	28,638.79	6,439.05	9,505.40	7,554.58	4,373.65
	Flaw material	28,962.79	29,489.48	31,118.97	26,945.02	27,746.08	14,852.69	13,905.27	7,381.29
	Biomass fuel	13,540.44	10,452.08	3,023.91	2,413.56	4,314.90	5,494.63	2,241.30	1,502.30
	Genetic resources	13,608.72	11,270.42	11,128.64	6,291.94	15,646.84	7,373.27	4,483.02	1,645.07
	Subtotal	**103,641.48**	**94,952.54**	**87,952.50**	**73,380.81**	**64,120.96**	**46,677.07**	**32,219.65**	**17,313.09**
Regulating services	Water regulation	16,203.23	15,162.46	14,726.35	14,406.67	804.05	586	3,470.95	3,219.50
	Water purification	39,522.41	34,136.74	18,432.00	9,685.05	21,420.05	14,935.33	11,725.10	7,539.19
	Erosion control	22,683.52	18,177.33	13,811.49	1,337.36	18,632.03	11190.0075	5,761.71	2,409.89
	Biological control	4,636.39	4,575.77	5,037.00	4,450.50	1,614.01	2178.45	1,350.57	752.6
	Biodiversity protection	19,186.24	16,236.89	14,273.65	4,970.70	13,721.37	8820.1	5,059.58	2,356.13

	Climate regulation	56,916.78	57,645.19	51,707.04	38,255.54	41,228.65	26,748.99	25,461.55	15,004.15
	Gas control	1,199.03	948.46	788.05	11.93	1,084.74	610.97	302.1592	110.6
	Carbon sequestration	87,029.15	70,799.38	58,866.47	15,411.06	63,347.48	42,752.57	20,439.32	8,390.55
	Subtotal	**247,376.75**	**217,682.22**	**177,642.05**	**88,528.81**	**161,852.37**	**107,822.41**	**73,570.95**	**39,782.61**
Supporting services	Nutrient recycling	481	398.22	393.23	-	554.3775	261.225	158.5926	58.05
	Pollination	5,929.17	5,180.11	4,721.42	3,026.40	2,867.41	2793.525	1,372.24	674.45
	Soil formation	955.88	662.93	223.26	34.65	238.475	402.35	44.5316	16.3
	Habitat/regulation	1,162.40	962.36	950.29	-	1,339.72	631.2825	383.25862	140.285
	Subtotal	**8,528.45**	**7,203.62**	**6,288.20**	**3,061.05**	**4,999.98**	**4,088.38**	**1,958.62**	**889.09**
Cultural services	Recreation	35,956.31	29,957.63	29,398.74	1,490.69	38,746.76	18386.47	11,510.24	4,474.17
	Tourism	3,055.05	2,529.28	2,497.57	-	3,521.09	1659.15	1,007.29	368.7
	Subtotal	**39,011.36**	**32,486.91**	**31,896.31**	**1,490.69**	**42,267.85**	**20,045.62**	**12,517.53**	**4,842.87**
	Total	**398,558.04**	**352,325.29**	**303,779.06**	**166,461.36**	**273,241.16**	**178,633.48**	**120,266.76**	**62,827.66**

2.7.4 Sensitivity analysis result on LULC

The CS value was less than one. According to the global ESV assessment, CS for Eucalyptus was higher at 0.38%, 0.43%, and 0.36% in the Legedadi watershed in 1985, 1995, and 2022, respectively. Whereas the CS for natural vegetation was higher at 0.5% in 2010. Similarly, according to the local ESV assessment, CS for Eucalyptus was higher in 1985 and 1995, at 0.29% and 0.32%, respectively, whereas CS for natural vegetation and cultivated land was higher in 2010 and 2022, at 0.31% and 0.33%, respectively (Table 11). According to the global ESV assessment, CS for natural vegetation was higher at 0.62%, 0.5%, and 0.46% in the Dire watershed in 1985, 1995, and 2022, respectively, whereas CS for Eucalyptus was higher at 0.6% in 2010. However, according to the local ESV assessment, CS for natural vegetation was higher in 1985 and 1995, at 0.59% and 0.41%, respectively, whereas CS for Eucalyptus was higher in 2010 and 2022, at 0.48% and 0.55%, respectively (Table 12). This is due to the large size of the land and the high ESV value assigned to the land use types.

Table 11: Change in estimated total ESV and coefficient of sensitivity (CS) after a 50% adjustment of ecosystem services valuation coefficient (VC) in the Legedadi watershed.

	1985		1995		2010		2022	
	%	CS	%	CS	%	CS	%	CS
Natural vegetation VC ± 50%	9.9	0.36	9.9	0.32	9.8	0.50	9.87	0.32
Eucalyptus plantation VC ± 50%	9.9	0.38	9.9	0.43	9.8	0.15	9.87	0.36
Cultivation VC ± 50%	4.72	0.00	4.71	0.02	4.71	0.09	4.7	0.11
Grassland VC ± 50%	6.71	0.12	6.7	0.09	6.69	0.03	6.69	0.01
Settlement VC ± 50%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Waterbody VC ± 50%	5.53	0.14	5.52	0.14	5.52	0.23	5.51	0.20
Bare VC ± 50%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 12: Change in estimated total ESV and coefficient of sensitivity (CS) after a 50% adjustment of ecosystem services valuation coefficient (VC) in the Dire watershed.

	1985		1995		2010		2022	
	%	CS	%	CS	%	CS	%	CS
Natural vegetation VC ± 50%	8.16	0.62	7.4	0.50	6.3	0.31	4.96	0.46
Eucalyptus plantation VC ± 50%	8.16	0.36	7.4	0.36	6.3	0.60	9.72	0.07
Cultivation VC ± 50%	3.88	0.01	3.64	0.01	4.23	0.01	8.37	0.15
Grassland VC ± 50%	5.52	0.02	5.01	0.13	0.00	0.00	0.00	0.00
Settlement VC ± 50%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Waterbody VC ± 50%	0.00	0.00	0.00	0.00	3.52	0.08	2.77	0.32
Bare VC ± 50%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

2.8 Discussions

Both watersheds are experiencing rapid changes in land use. Cultivation and settlement are rapidly encroaching on land use, displacing natural vegetation and grassland. Awraris (2017) reported a similar finding, that cultivated land dominated land use in the study watersheds. Natural vegetation has been rapidly lost, resulting in increased soil loss in both watersheds (AAWSA, 2016; Awraris, 2017). As a result, the Legedadi reservoir has experienced severe siltation, with an average annual siltation rate of 110,500 m^3, and its volume capacity decreased by 2.1 MCM between 1979 and 1998 (AAWSA, 2011). The annual soil loss in the Legedadi watershed was estimated between 69.9 and 138.4 tons/ha (Gebreegziabher et al., 2021).

In the Ledegadi watershed, population density increased by 65% in just one decade (AAWSA, 2011; 2016). Rapid population growth and radical outward settlement expansion are endangering Addis Ababa's and surrounding areas' water sources (Tamiru et al., 2005).

Thus, rapid population growth (due to increased natural birth rate and rural-urban migration) is expected to increase the demand for farmlands and settlement, destroying natural areas and worsening watershed capacity in generating vital ecosystem services, particularly hydrological services. Water users in Addis Ababa have been extremely dissatisfied with the current water supply, which has decreased in quantity and quality over time (Zinabu and Michael, 2020). Furthermore, the city's unmet water demand is expected to rise to 307 MCM by 2030 (Zinabu and Michael, 2020).

The overall ESVs of the watersheds have decreased dramatically during the study period because of the loss of important natural habitats, particularly natural forest, and grassland habitats. The primary contributor to the decline of ESVs in the watersheds was the dramatic conversion of natural vegetation and grassland to settlement and cultivation (Kindu et al., (2016); Terefe et al., (2016); Terefe et al., (2017); Hailu et al., (2019); Abreham et al., (2019); Hailu et al., (2021) reported a similar dramatic reduction in ESVs in various parts of Ethiopia. Terefe et al. (2017) estimated a total ESV decline of approximately US$ 3.69 million between 1973 and 2015 in Ethiopia's Chillimo forest central highlands. Likewise, due to the expansion of cultivation, settlement, and development of the Dire reservoir, the natural habitat particularly the grassland in the Dire watershed has greatly deteriorated, thus a substantial reduction in ESV was obtained.

In general, all ecosystem services declined across all periods in both watersheds, except food production; the increased cropland size and a higher ecosystem service value attributed to it have caused an increase in food provisioning service (Tatek, 2022). Although the construction of the Dire water reservoir in 1999 contributed to the increase of water regulation services between 2010 and 2022; further, land use modifications could have a substantial negative impact on the watershed's hydrological services and affect people's welfare (Yilikal et al., 2019). Moreover, greatly increased soil erosion and consequently resulted in a substantial sediment load to the water reservoirs. To reverse the adverse impact of land use changes a quick management intervention is required such as the introduction of agroforestry practices in the entire watershed system, a considerable amount of land is suitable for apple-based farming practices, which could meaningfully restore watershed ecosystems and enhance local income (Simeneh et al., 2022).

2.9 Conclusion

The findings of the study revealed that the replacement of natural vegetation by settlement and farmland was highly dynamic. The anthropogenic disturbance harmed the ecosystem process, resulting in a decrease in ESV in both watersheds. The decline in ESV is expected to continue as both watersheds experience rapid land conversion. To address the complex socio-ecological problems in the watersheds, an urgent and active landscape restoration scheme is required with the active participation of key stakeholders. To ensure effective and sustainable watershed management intervention; the use of innovative market-based sustainable land management practices can be a valuable tool for managing the ecosystems and improving the livelihoods of nearby farmers. However, further research on the prospects of economic instruments needed to be assessed.

Chapter Three

3 Assessing Habitat Quality using the InVEST model in the Dire and Legedadi watersheds.

3.1 Abstract

This study aimed to quantify and map habitat quality in the Dire and Legedadi watersheds between 1985 and 2022 using the InVEST habitat quality model. The purpose of this study was to assess the spatiotemporal changes in habitat quality in the Dire and Legedadi watersheds. The result showed that between 1985 and 2022, both watersheds experienced a substantial decline in habitat quality. The extent of high habitat quality habitat in the Legadadi watershed has decreased considerably from 206 km^2 in 1985 to 50.26 km^2 in 2022. Similarly, high habitat quality habitat in the Dire watershed has gradually declined from 87.29 km^2 in 1985 to 35.44 km^2 in 2022. The construction of the Dire water reservoir in 1999 helped to maintain the extent of moderate habitat quality which increased from 6.93 km^2 in 1995 to 25.36 km^2 in 2010. The rapid expansion of farmland and settlement at the expense of natural vegetation was strongly linked to the decline in habitat quality. As a result, appropriate watershed management intervention is required. Therefore, the outcome of this research will provide a scientific basis for future ecosystem monitoring to monitor the impacts of watershed management interventions.

Keywords: Dire and Legedadi, Watersheds, InVEST, Habitat quality,

3.2 Introduction

A high-biodiversity ecosystem is regarded as a high-quality habitat that contributes to improved ecosystem processes (Johnson, 2007). The decline in biodiversity resources and intactness, on the other hand, is having a significant negative impact on habitat quality because of several unprecedented LULC changes such as increased settlement, farming, and artificial monoculture plantations (Abreham et al., 2020; Zhang et al., 2020; Hamere et al., 2021). The degradation of habitat quality was primarily caused by intensive land use expansion associated with human activities (Liang and Liu, 2017). As a result, the proximity to and intensity of adverse land uses has a significant impact on ecosystem functioning and process (Fu et al., 2013; Yan et al., 2018), and the ability to generate ecosystem services can deteriorate (Polasky et al., al., 2011; Villamagna et al., 2013; Thomas et al., 2021).

Similarly, the study watersheds have drastically changed interms of LULC (Simeneh et al., 2023a). Natural forests and grassland were the dominant land cover types until the late 1980s when they were largely replaced by farmland and settlement (AAWSA, 2016; Awraris, 2017). Due to the increased deterioration of natural habitats in both watersheds, the water quality in the reservoirs was reported to be very low (Taye, 2009; Helnata, 2019). According to a recent study, planting an apple tree can help restore watershed ecosystems and increase local income (Simeneh et al., 2022). Understanding landscape patterns, the interaction of human activities, and the effects of land conversion on natural phenomena is critical for proper land management and decision-making that contributes to the preservation of ecosystem integrity (Rawat and Kumar, 2015). The study of watersheds' rapid LULC changes and their effects on the quality and quantity of ecosystem services are poorly known and less understood. Assessing the effects of LULC changes on habitat quality and ESVs in the Dire and Legedadi watersheds can help with proper ecosystem management intervention, which can help humans meet a variety of economic and environmental needs.

Assessments of habitat quality and ecological risk are required as part of sustainable ecosystem management to reduce the negative effects of anthropogenic disturbances (Zhang et al., 2022). This study's assessment model can be used as an effective decision-support tool to prioritize important ecological areas and can also be adapted for use in ecosystem management intervention (Zhang et al., 2022). The availability of biodiversity resources in a specific environment and the environment's ability to provide suitable living conditions are influenced by quality habitat (Hall et al., 1997). It is regarded as a critical representation of regional biodiversity and ecosystems, as well as a critical link in ensuring regional ecological security and enhancing human well-being (Chen et al., 2016). Understanding the temporal and geographical aspects of habitat quality is the foundation for developing land-use planning and management (Liu et al., 2018).

To quantify habitat quality, the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) model has been widely used (Sharp et al., 2020; Abreham et al., 2020; Zhang et al., 2020; Hamere et al., 2021). The model provides an excellent research method and perspectives (Terrado et al., 2016; Romero and Luque, 2006), as well as information on land use and threats to computing habitat quality. The spatial extent of habitat quality within the landscape was determined by the habitat's proximity to human-dominated land use and the intensity of disturbance caused by the land use (Sharp et al., 2020). We hypothesize that habitat quality deteriorated over time because of threat factors present in the study area. As a result, the purpose of this study was to assess the spatiotemporal changes in habitat quality in the Dire and Legedadi watersheds using InVEST software to provide scientific foundations for sustainable watershed management practices in Addis Ababa’s rapidly deteriorating headwaters.

3.3 Materials and Methods

3.3.1 The InVEST Habitat Quality Model.

The InVEST model offers a good research method as well as perspectives (Hamere et al., 2021; Abreham et al., 2020; Sharp et al., 2020; Zhang et al., 2020; Terrado et al., 2016; Romero and Luque, 2006). To compute habitat quality, the model incorporates information on land use and threats. The spatial extent of habitat quality within the landscape was determined by the habitat's proximity to human-dominated land use and the intensity of disturbance caused by the land use (Sharp et al., 2020). The model considers LULC with higher habitat quality to be relatively intact and capable of supporting increased biodiversity, whereas LULC with lower habitat quality indicates reduced biodiversity support and denotes a degraded habitat (Baral et al., 2014). The model is based on the relative impact of threats to habitats, the distance between the threat source and the habitat, and the sensitivities of specific habitats to any potential threats that could lead to habitat degradation (Chen et al., 2016; Sharp et al., 2020).

For this study, five biodiversity threats were identified, namely settlement, farming, eucalyptus dominance, road, and land degradation, and the significance (weight) of each threat was prioritized based on the opinions of local key informants, experts' knowledge, and review of the literature (Abreham et al., 2020; Hamere et al., 2021), using the approach developed by Terrado et al., (2016) and Wu et al (2014) (Table 13).

The key inputs of habitat quality in the InVEST model, according to Sharp et al., 2020, are the suitability of each LU/LC type (Hj) for providing habitat for biodiversity; second, anthropogenic threats that originate at pixel x (rx) affecting habitat quality; and third, the sensitivity of each LU/LC type to each threat (Table 13).

$$i_{rxy} = 1 - \left(\frac{d_{xy}}{d_{rmax}}\right) \quad \text{……………………………if linear……….5}$$

$$i_{rxy} = \exp\left(-\left(\frac{2.99}{d_{rmx}}\right)d_{xy}\right) \quad \text{……………………………if exponential……6}$$

Where, d_{xy}= linear distance between grid cells x and y, and d_{rmax} = maximum effective distance of threats r's across space (Sharp et al., 2020).

The total threat level in a grid cell x with LULCj is calculated as the relative habitat suitability score (Hj), which ranges from 0 to 1, with 1 indicating the highest suitability to species (Sharp et al., 2020). The model's final input is the sensitivity of habitat type to different threats; this allows for the differentiated impacts of threats on different habitats to be accounted for. Threat impacts on habitat are determined by 1) the threat's effect over space (irxy); 2) the relative weight of each threat's importance compared to the others (wr); and 3) the relative sensitivity of each habitat to the respective threat (Sjr) (Sharp et al., 2020). The grid x's stress level Dxj with land-use type j is calculated as follows.

$$D_{jx} = \sum_{r=1}^{R} \sum_{y=1}^{Yr} \left(\frac{w_r}{\sum_{r=1}^{R} w_r}\right) r_x i_{rxy} \theta_x S_{jr} \text{…………7}$$

where R= number of threat factors, y_r = set of grid cells on r's map, w_r =relative effect of each threat, θ_x = level of accessibility to a grid cell x, and S_{jr} = relative sensitivity of each habitat type to each threat (Sharp et al., 2020).

The model's output ranges from 0 to 1, with 1 representing the highest level of habitat quality (Sharp et al., 2020). Threats with higher destructive values (on a scale of 0-1) have higher impacts on habitat, and the more sensitive a habitat type is to a threat (higher Sjr), the more degraded the habitat type could be by the threat.

The environmental level that the ecological environment provides for the survival of individual organisms and populations is referred to as habitat quality (Yanan et al., 2022). It is a variable with a numerical range ranging from low to high (Sharp et al., 2020; Yanan et al., 2022). The higher the quality of the habitat, the more stable the ecosystem's ecological structure and function (Sharp et al., 2020; Yanan et al., 2022). The manner and intensity with which human land use determines habitat quality, and the more intense the land use, the more pronounced the decline in habitat quality (Almpanidou et al., 2014). The degree of habitat degradation was used to calculate habitat quality, and the habitat quality score decreased as the degree of habitat degradation increased (Sharp et al., 2020; Yanan et al., 2022). The habitat quality calculation formula is as follows (Sharp et al., 2020).

$$Q_{xj} = H_j \left[1 - \left(\frac{D_{xj}^2}{D_{xj}^2 + k^z}\right)\right] \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots 8$$

where, Qxj is the habitat quality of grid cell x in land cover type j; Hj is the habitat suitability of land cover type j; Dz xj is the level of habitat threat for grid cell x in land cover type j; k is the half-saturation factor, which is generally taken as half of the maximum value of Dz xj; and x is a constant.

Table 13: Ecological habitat quality input data used for the InVEST habitat quality model in the Dire and Legedadi watersheds.

Threat	Maximum Distance	Weight	Decay	LULC types						
				NV	W	S	GL	EP	C	BL
				Habitat suitability score						
				1	1	0	1	1	0	0
				Habitat sensitivity to threats						
Eucalyptus dominance	1	0.75	Linear	0.75	0.5	0.2	0.8	0.00	0.00	0.00
Settlement	2	1.00	Exponential	1.00	0.8	0.00	1.00	0.8	0.00	0.6
Farming	2	1.00	Exponential	1.00	1.00	0.00	1.00	0.8	0.00	0.00
Roads	3	0.75	Linear	0.8	0.75	0.00	0.5	0.5	0.2	0.5
Land degradation	1	0.60	Linear	0.6	0.6	0.2	0.6	0.6	0.6	0.6

3.3.2 Data Preparation and Input for the InVEST- Habitat Quality Model.

The InVEST habitat quality model requires data inputs (both spatial and non-spatial) (Figure 5). Thus, LU/LC maps, threat sources, and impacts, habitat types, habitat sensitivity to each threat, and half-saturation constant were all required inputs (Sharp et al., 2020). The information on LULC was obtained from Simeneh's (2023a) previous study in the study area. To run the habitat quality model, all the required inputs were loaded, including LULC maps for the respective years, threat sources and impacts, habitat types, and habitat sensitivity to each threat. Finally, habitat quality maps for each year were created; the final habitat quality maps were classified into three categories (low, moderate, and high).

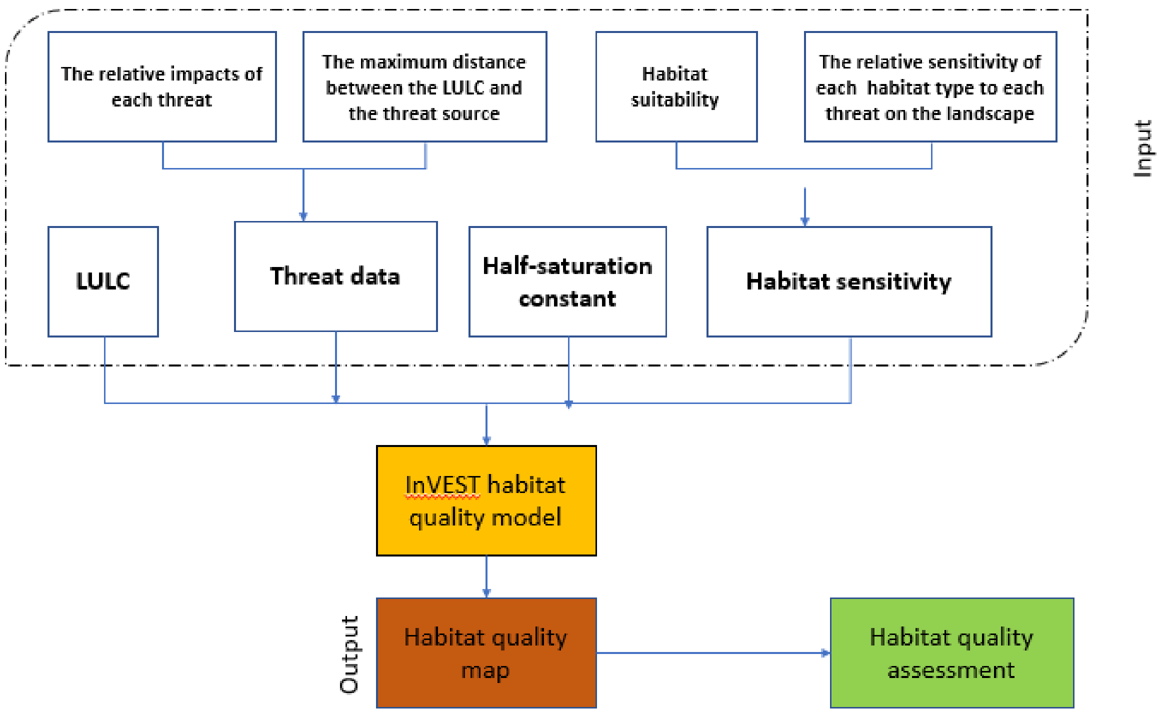

Figure 5: Flowchart showing methodological steps followed in the study.

3.4 Results and Discussions

Between 1985 and 2010, the watersheds were dominated by high habitat quality habitats (Figure 6; 7). In the Dire watershed, the extent of high habitat quality has gradually declined between 1985 and 2022, from 87.29 km^2 (89.84%) in 1985 to 78.23 km^2 (80.56%) in 1995, 71.81 km^2 (73.9%) in 2010, and 35.44 km^2 (36.48%) in 2022. Low habitat quality, on the other hand, has increased from 9.87 km^2 (10.16%) in 1985 to 11.95 km^2 (12.31%) in 1995 and then to 49 km^2 (50.44%) in 2022. (Table 14). The construction of the Dire water reservoir in 1999 helped to maintain the extent of moderate habitat quality. It increased from 6.93 km^2 (7.14%) in 1995 to 25.36 km^2 (26.1%) in 2010 but then decreased to 12.71 km^2 (13.08%) in n 2022 (Table 14; Figure 8).

The extent of high habitat quality habitat in the Legadadi watershed has decreased extensively from 206 km^2 (95.5%) in 1985 to 179.5 km^2 (83.2%) in 1995 and then to 156.85 km^2 (72.66%) in 2010. However, by 2022, the high habitat quality would have dropped to 50.26 km^2 (23.28%). Low habitat quality, on the other hand, increased from 9.62 km^2 (4.46%) in 1985 to 36.31 km^2 (16.82%) in 1995, then to 59 km^2 (27.34%) in 2010 and 165 km^2 (76.72%) in 2022. (Table 14; Figure 9).

In the last 37 years, the quality of the habitat has deteriorated dramatically. In general, high-quality habitats in 1985 degraded into low-quality habitats by 2022 (Table 14). This is due to an unprecedented increase in anthropogenic pressure, which has resulted in severe changes in the ecosystem's healthy functioning. A similar finding was reported that only 1.96 % of the area has high environmental quality; the medium, poor extremely deteriorated quality environment consisted of 77%, 13.30%, and 8.12%, respectively (AAWSA, 2011). Rapid anthropogenic LULC changes associated with the expansion of human settlements and artificial plantations may result in a significant loss of habitat quality (Jie et al., 2015; Li et al., 2018; Liting et al., 2019; Abreham et al., 2020; Hamere et al., 2021) and pose a threat to biodiversity (Sun et al., 2019; Kunwar et al., 2020). Similarly, rapid urbanization and agricultural expansion have destroyed previously dominant natural habitats, resulting in substantial habitat quality degradation in watersheds. Natural habitat loss can have a variety of effects on the larger ecosystem and have a significant impact on human well-being. The magnitude of poor habitat quality has grown considerably as settlement and cultivation areas have expanded. With a declining trend, high habitat quality values have only occurred in areas where natural habitats and reservoirs are located. Human activities, particularly settlement and urbanization, unrestricted grazing, deforestation, and agricultural practices, have the potential to degrade habitat quality and disrupt ecological processes. Similar findings have been reported in the Winkie and Beressa watersheds (Abraham et al., 2020; Hamere et al., 2021). Furthermore, eucalyptus plantations' dominance and land degradation have contributed to the watershed's declining habitat quality (Hamere et al., 2021).

Table 14: Habitat quality changes in the Legedadi and Dire watersheds using the InVEST habitat quality model (1995, 2022)

Habitat quality	Legadadi watershed							
	1985 (km^2)	%	1995 (km^2)	%	2010 (km^2)	%	2022 (km^2)	%
Low	9.62	4.46	36.31	16.82	59.03	27.34	165.62	76.72
Moderate	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
High	206.27	95.54	179.57	83.18	156.85	72.66	50.26	23.28
Total	215.89	100.00	215.88	100.00	215.88	100.00	215.88	100.00
	Dire watershed							
Low	9.87	10.16	11.95	12.31	0.00	0.00	49.00	50.44

Moderate	0.00	0.00	6.93	7.14	25.36	26.10	12.71	13.08
High	87.29	89.84	78.23	80.56	71.81	73.90	35.44	36.48
Total	97.16	100.00	97.11	100.00	97.17	100.00	97.15	100.00

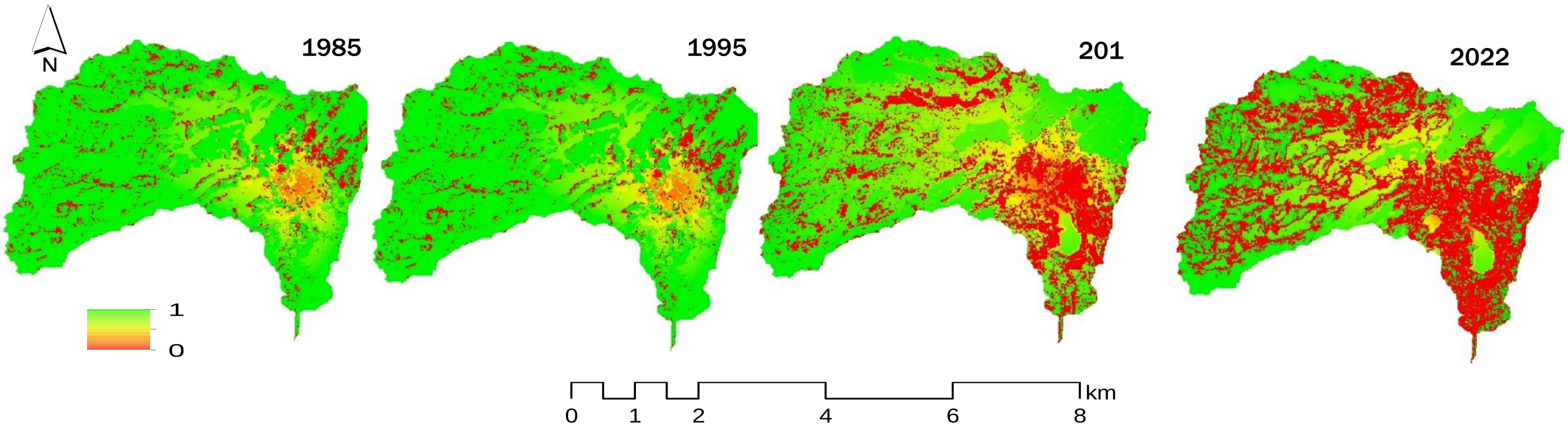

Figure 6: Habitat quality in the Dire watershed (1985-2022)

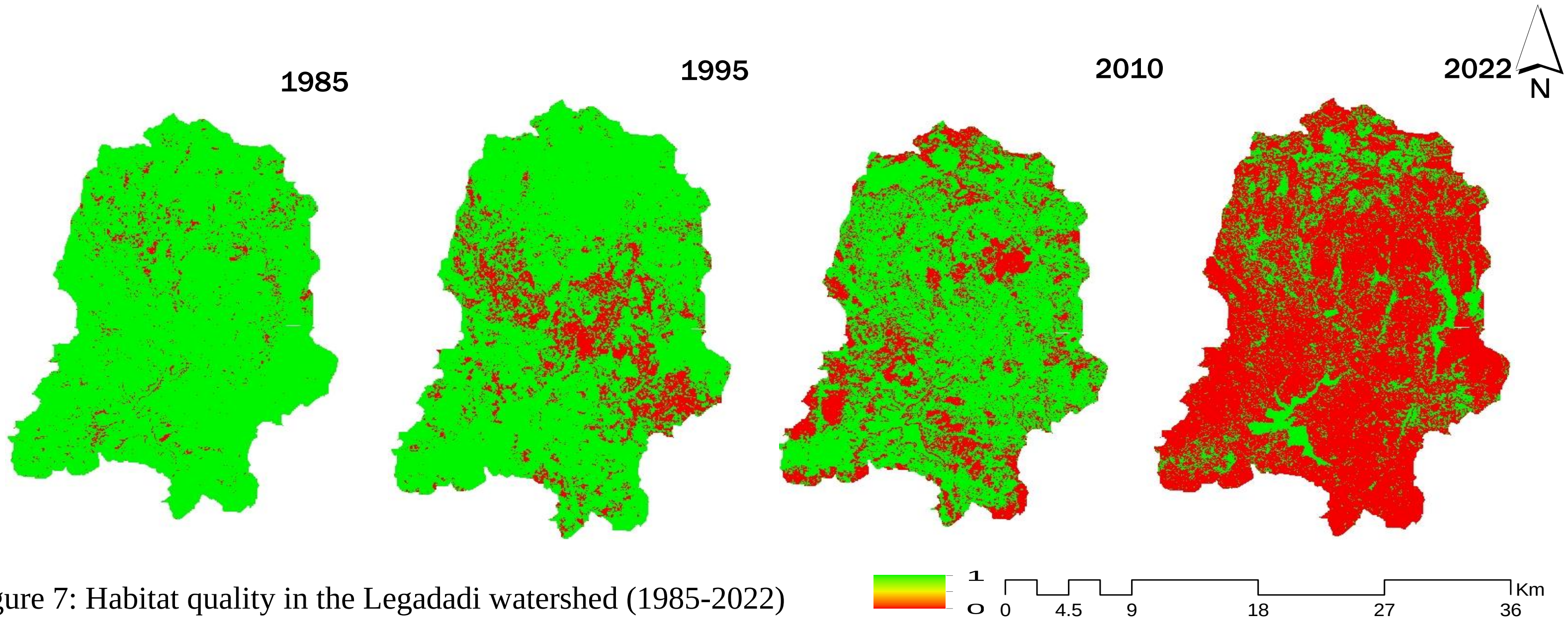

Figure 7: Habitat quality in the Legadadi watershed (1985-2022)

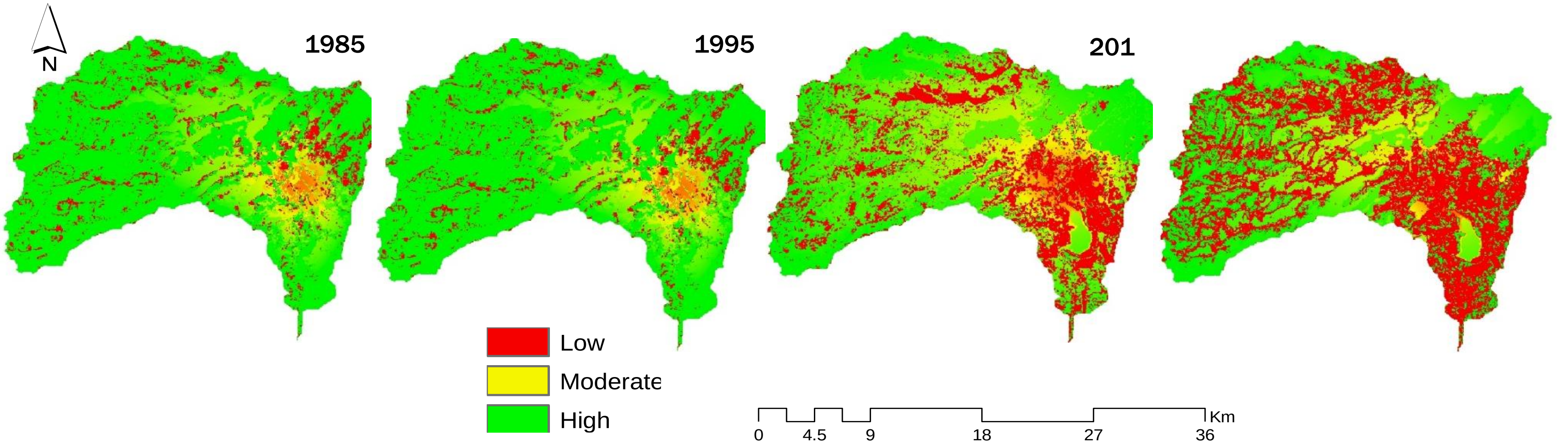

Figure 8: Classified habitat quality maps in the Dire watershed

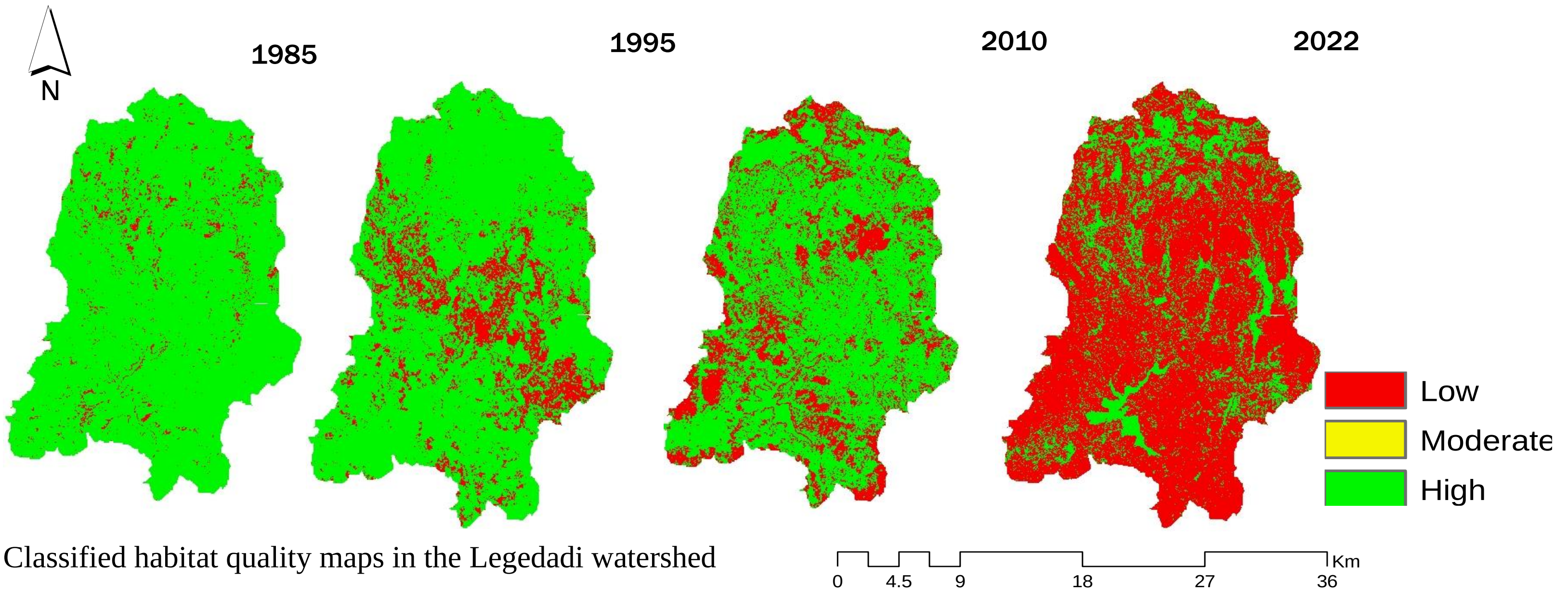

Figure 9: Classified habitat quality maps in the Legedadi watershed

3.5 Conclusion

By mapping critical habitats for ecosystem conservation, habitat quality modeling aids in sustainable land management initiatives. The rapid expansion of farmland and settlement in the study watersheds has largely resulted in a decline in habitat quality. The decline in habitat quality is expected to continue. To improve habitat quality in both watersheds, a sustainable water management strategy should be implemented based on the severity and extent of habitat quality degradation. It is also critical to ensure that the watershed management intervention causes little or no economic loss to other land uses (mainly for agriculture) through implementing appropriate incentive-based watershed management practices in the study landscapes. Thus, this study will assist decision-makers in appropriately planning for sustainable watershed management practices in the study watersheds.

Chapter Four

4 Impacts of Land Use Land Cover Changes and Climate Variability on Water Yield in the Central Watersheds of Ethiopia

4.1 Abstract

Water yield capacity assessment is critical for water management and ecosystem improvement. Using the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) model, we assessed the effects of land use changes and climate variability on annual water yield in the Dire and Legedadi watersheds. The model was run using meteorological data, land use, soil depth, plant water content, and biophysical parameters. The impact of land use changes and climate variability was investigated by creating three scenarios: actual conditions, actual conditions with Land Use and Land Cover (LULC) remaining constant, and actual conditions without climate variability. The findings revealed that the total water yield in both watersheds has increased. The Legedadi watershed increased total water yield by 15.32% from 111.6 million m^3 (1149 millimeters) in 1995 to 131.8 million m^3(1357.5 millimeters) in 2021; while the Dire watershed increased total water yield by 32.5% from 259.5 million m^3(1202 millimeters) in 1995 to 386.6 million m^3(1790 millimeters) in 2021. The effect of climate variability on annual water yield was approximately 99.9% and 73.3% in the Legedadi and Dire watersheds, respectively; land use change was 0.01% and 26.7%. Despite its higher water yield, urban and agricultural land expansion may have an impact on water yield. As a result, participatory watershed management interventions that consider landscape patterns are required to optimize and maintain ecosystem services.

Keywords: Ecosystem services, InVEST, Annual Water Production, Landscape

4.2 Introduction

Water yield is an important indicator of ecosystem function and plays an important role in sustainable development (Peijie et al., 2021). It has a direct impact on key sectors such as agriculture, industry, fisheries, and domestic activities and is critical for socio-economic development (Pessacg et al., 2015; Bower, 2014; Elbeltagi et al., 2020; Yang and De Hua, 2021). Understanding changes in watersheds requires an assessment of the spatiotemporal variation of water yield (Lu et al., 2020). The collective effects of climate and land use changes can have an impact on watershed capacity (Yu et al., 2015; Peijie et al., 2021). Climate change has an impact on water production capacity by altering the pattern and intensity of precipitation and temperature (Lian et al., 2020; Dagnachew et al., 2003; Song *et al.,* 2015). While the impact of land use changes can vary (Yue et al., 2018; Zhang et al., 2018), for example, natural area deterioration has a significant impact on watershed hydrologic cycles, affecting evapotranspiration (Ajanaw, 2021), infiltration, and water retention rates, as well as changing the rainfall pattern and intensity (World Commission on Dams 2000; Ennaanay, 2006; Bonan, 2015; Srivastava et al., 2020; Aghsaei et al., 2020).

Ethiopia has been designated as a disaster-prone country, and its people are particularly vulnerable to climate change. Climate erraticism has resulted in increased temperature extremes, erratic rainfall, and recurrent severe flooding during the main rainy season, displacing thousands of people in various parts of Ethiopia. The Dire and Legedadi watersheds are important headwaters for the Blue Nile and Awash River basins, as well as the main water sources for Addis Ababa. As a result, hydrological stability and water supply service function in these watersheds are critical for socioeconomic development. However, they are facing severe environmental issues because of unprecedented anthropogenic pressures. The conversion of natural forests and grassland into farmland and settlement (AAWSA, 2016; Awraris, 2017; Simeneh et al., 2023a) has impacted habitat quality and overall ecosystem services in the Dire and Legedadi watersheds (Simeneh et al.,2023b). Such changes may have a negative impact on watershed hydrological services, reducing water quality and quantity (Ajanaw, 2021). As a result, downstream water users in Addis Ababa have been extremely dissatisfied with the water supply, as both the quantity and quality of water have decreased over time (Yilikal et al., 2019).

Several studies have used the InVEST model to assess landscape water production capacity and determine the impacts of climate and land use changes, including Pessacg et al., (2015) in the Chubut River Basin; Sangam et al., (2016) in the Bago River Basin; Lang et al., (2017) in the Sancha River Basin; Goyal and Khan, (2017) in the Sutlej River Basin; Bai et al., (2019) in Kentucky; Mesfin *et al.*, (2019) in the Wabe catchment; Yang *et al.*,(2019) in the monsoon catchments; Aghsaei *et al.*, (2020) in the Anzali wetland catchment; Guodong *et al.*,(2020) in North China; Hu *et al.*,(2020) in the Dongting Lake wetland region; Mo *et al.*, (2021) in the Dongjiang Lake Basin; Peijie *et al.*, (2021) in the Shule River Basin; and Ningrum *et al.*,(2022) in Tesso Nilo National Park. Understanding the water services provided by watersheds could aid in addressing the city of Addis Ababa's and its surrounding areas' water scarcity problem. Several assessments have been conducted, with the primary focus on monitoring changes in water quality and quantity in the water reservoirs (Getahun, 2014; Sisay, 2014; Yilikal et al., 2018), but there has been little in-depth investigation of the watershed's ecosystem processes and landscape patterns. As a result, the goal of this study was to assess the landscape's annual water production capacity and estimate the effects of changes in land use and climate on the water production capacity of the Dire and Legedadi watersheds.

4.3 Materials and Methods

4.3.1 Data sources and preparation

Meteorological data, land use, soil depth, plant available water content, watersheds, and the biophysical table are all used to run the water yield model. Meteorological and land use data were collected for the years 1995, 2010, and 2021. The National Meteorological Agency (NMA) provided meteorological data from stations (Sendafa, Aletu, Dire Gidib, Entoto, and Sulutla). The average value of the nearest stations was used to replace the missing methodological data (Ferrari and Ozaki, 2014; Mohammad et al., 2017).

The meteorological data were then spatially interpolated to the entire watersheds using the geographically weighted regression Kriging algorithm in the ArcGIS 10.5 spatial analyst tool. Counting the number of rainy days in each month yielded the number of rain events for each year. The annual potential evapotranspiration for the Legedadi watershed was obtained from Habtamu (2017) and then spatially interpolated into 1 km × 1 km grid pixels using the CROPWAT 8.0 software for the Dire watershed (Zotarelli et al., 2015). Based on the physical and chemical properties of the soil, the plant available water capacity (PAWC), which is the difference between the wilting point and field capacity can be estimated (Zhou et al., 2005; Easton, 2016). Field capacity is the amount of water that remains in the soil profile after 48-72 hours of free drainage after saturated conditions, whereas the wilting point is the minimum moisture content of the soil that can support plant growth and below which plants cannot recover (Easton, 2016) (Table 15).

Table 15: Average available water content for various soil textural classes (Easton, 2016)

Soil textural class	Wilting point	Field capacity	Available water
	% Moisture		
Sand	5	12	7
Sandy loam	9	21	12
Loam	16	36	20
Silt loam	18	39	21
Clay loam	24	39	15
Silty clay	24	39	13
Clay	27	39	12

Soil data were obtained from (Sisay, 2014; Simeneh et al., 2022); the fraction of plant available water content was calculated by calculating the wilting point and field capacity of the various soil types in the study watersheds. The crop coefficients (Kc) for each LULC were obtained from the InVEST user's guide (Sharp et al., 2020), (Table 16) and the study on LULC changes in watersheds conducted by Simeneh et al., (2023a) was used (In Figure 3; 4). The meteorological data obtained from the NMA was examined. In 1995, 2010, and 2021, the mean maximum temperature in Legedadi was 23.5°C, 22.6°C, and 22.9°C, respectively, while it was 23.5°C, 23.6°C, and 22.6°C in the Dire watershed. The Legedadi watershed had mean minimum temperatures of 2.9°C, 9.3°C, and 7.4°C in 1995, 2010, and 2021, respectively, while the Dire watershed had mean minimum temperatures of 2.9°C, 9.7°C, and 8.8°C in 1995, 2010, and 2021, respectively. Annual precipitation in the Legedadi watershed was 1302 mm, 1108 mm, and 1887.2 mm in 1995, 2010, and 2021, respectively, while in the Dire watershed, it was 1220 mm, 1295.8 mm, and 1409.5 mm in 1995, 2010, and 2021, respectively.

Table 16: Kc for different land cover types (Sharp *et al.*, 2020).

Land cover	Kc
Irrigated cropland	1.2
Field cropland/ Forest closed/ Stream/Flooded/marsh	1
Forest	0.9
Natural shrub	0.8
Urban non-vegetated	0.1
Residential	0.1

4.3.2 Water Yield Model InVEST

The InVEST water yield model has been tested, and reliable results have been obtained (Guodong et al., 2020; Peijie et al., 2021). The model quantifies the relative water contributions of different parts of a landscape, and it also provides insight into how changes in land use patterns affect annual surface water and hydropower production (Sharp et al., 2020). The model assumes the amount of water runoff from the watershed to determine the water yield, i.e., the removal of precipitation from the water storage and evapotranspiration losses (Sharp et al., 2020).

The model runs on a gridded map, and the pixel-scale calculations allow for the representation of the heterogeneity of key water yield driving factors such as soil type, precipitation, and vegetation type (Sharp et al., 2020). In every 300 m by 300 m grid cell, the average annual precipitation (Px), annual reference evapotranspiration, soil depth, plant available water content, plant root depth, and land use characteristics are used to calculate the average annual water yield (Yxj) as follows (Sharp et al., 2020).

$$\mathrm{Yxj} = 1 - \left(\frac{\mathrm{AETxj}}{\mathrm{Px}}\right) * \mathrm{Px} \ldots\ldots\ldots\ldots\ldots\ldots\ldots 8$$

Where AET is the annual actual evapotranspiration and AETxj, Px is an approximation of the Budyko curve (Zhang et al., 2001) given as:

$$\frac{\mathrm{AWCx}}{\mathrm{Px}} = \frac{1+\mathrm{PETx}}{\mathrm{Px}} - [1 + \left(\frac{\mathrm{PETx}}{\mathrm{Px}}\right)^{\omega}]^{\frac{1}{\omega}} \ldots\ldots\ldots\ldots 9$$

Where PET (x) is the potential evapotranspiration and ω (x) is a non-physical parameter that characterizes the natural climatic-soil properties. The Potential evapotranspiration PET (x) is defined as:

$$\mathrm{PET(x)} = \frac{\mathrm{Kc}(\ell \mathrm{x}) * \mathrm{ET0(x)}}{\mathrm{Px}} \ldots\ldots\ldots\ldots\ldots\ldots\ldots 10$$

Where ET0 (x) is the reference evapotranspiration from pixel x and (ℓx) is the plant (vegetation) evapotranspiration coefficient associated with the LU/LC ℓx on pixel x.

ET0 (x) reflects local climatic conditions, based on the evapotranspiration of reference vegetation grown in that location. (ℓx) is largely determined by the vegetative characteristics of the land use and land cover found on that pixel (Allen *et al.*, 1998; Allen *et al.*,2005). Kc adjusts the ET0 values to the crop or vegetation type in each pixel of the land use/land cover map. Ω (x) characterize the natural climatic-soil properties. It is an empirical parameter that can be expressed as follows Donohue *et al.*, (2012):

$$\omega(x) = Z(\frac{AWCx}{Px}) + 1.25 \ldots\ldots\ldots\ldots\ldots 11$$

Where AWC(x) is estimated as the product of plant available water capacity (PAWC) on pixel x, the minimum of root restricting layer depth and vegetation rooting depth (Sharp et al., 2020). Px is the annual precipitation on pixel x. (Z = 0.2 × N). Parameter Z, as a constant, represents the precipitation characteristics with a value from 1 to 30 and it is larger when the rainfall events are more frequent. Where N is the number of rainfall events (N > 1 mm) per year (Donohue et al.,2012; Redhead et al., 2016).

4.4 Scenarios of Climate Variability and LU/LC Change

It is critical to investigate the impact of land use changes and climate variability on water yield (Shilong et al., 2010). As a result, the annual water yield capacity of the watersheds (1995, 2010, and 2021) was estimated using actual conditions; then, the effects of climate variability and land use changes on water yield variations were analyzed using three explicit scenarios: actual conditions, actual conditions without LULC change, and actual conditions without climate variability. Under the actual scenario, LULC data and climate data were used in the model. Precipitation in 2010 and 2021 remains unchanged from 1995 in the absence of climate variability, and water yields for 1995, 2010, and 2021 were quantified to assess the impact of land use changes. The land use condition in 2010 and 2021 remains unchanged from 1995 in the scenario without land use change, and water yields were computed to determine the influence of climate variability (Table 17). Finally, the three scenarios were compared to see how climate variability and LULC changes affected water yield. Equations (12) and (13) (Lang et al., 2017) quantified the contribution of climate and LULC changes to water yield variability:

$$Gc = \frac{C}{C+L} * 100 \ldots\ldots\ldots\ldots\ldots\ldots 12$$

$$Gl = \frac{L}{C+L} * 100 \ldots\ldots\ldots\ldots\ldots\ldots 13$$

Where Gc is the contribution of climate variability to the change in water yield under the scenario without LULC change, GL is the contribution of LULC to the change in water yield under the scenario without climate change, C represents the difference in mean annual water yield in 2010 and 2021 under the scenario without LULC change, and L represents the difference in mean annual water yield in 2010 and 2021under the scenario without climate change.

Table 17: Simulations of water yield variation by scenario.

Year	**Scenarios**		
	Actual conditions	**Conditions without Climate variability**	**Conditions without Land Use Change**
1995	1995 precipitation	1995 precipitation	1995 precipitation
	1995 land use	1995 land use	1995 land use
2011	2010 precipitation	1995 precipitation	2010 precipitation
	2010 land use	2010 land use	1995 land use
2021	2021 precipitation	1995 precipitation	2021 precipitation
	2021 land use	2021 land use	1995 land use

4.5 Result

4.5.1 Annual water yield

The total water yield in both watersheds increased from approximately 371.1 MCM in 1995 to approximately 518.4 MCM in 2021. In 1995, the total water yield in the Legedadi was approximately 259.5 MCM, with a mean water yield (MWY) of 1202 millimeters (mm). It decreased to approximately 219.9 MCM in 2010 with a mean water yield (MWY) of 1018.6 mm, then increased to approximately 386.6 MCM in 2021 with a mean water yield (MWY) of 1790 mm (Table 18; Figure 10). The decrease in total water yield in 2010 could be attributed to lower annual rainfall in that year (Figure 10). Similarly, Dire's total water yield in 1995 was approximately 111.6 MCM, with a mean water yield (MWY) of 1149.5 mm. It increased to approximately 116.8 MCM with a mean water yield (MWY) of 1202 mm in 2010 and then improved to approximately 131.8 MCM with an MWY of 1357.5 mm in 2021 (Table 18; Figure 10).

The Legedadi watershed contributes the most to total water yield, accounting for approximately 70%, 65.5%, and 75% of total water yield in 1995, 2010, and 2021, respectively. Between 1995 and 2021, the total water yield in Legedadi increased by 32.5%, and it increased by 43% between 2010 and 2021, while it decreased by 18.9% between 1995 and 2010. Similarly, the total water yield in Dire has increased by 15.32% between 1995 and 2021, 11.38% between 2010 and 2021, and 4.45% between 1995 and 2010.

Table 18: Total and mean water yield characteristics in the Dire and Legedadi watersheds

Watershed	**Total Water Yield (10^6 m^3/year) (Water Yield Depth) (mm**		
	1995	**2010**	**2021**
Legedadi	259.5 (1202)	219.9 (1018.6)	386.6 (1790)
Dire	111.6 (1149.5)	116.8 (1202)	131.8 (1357.5)
Total	**371.1**	**335.8**	**518.4**

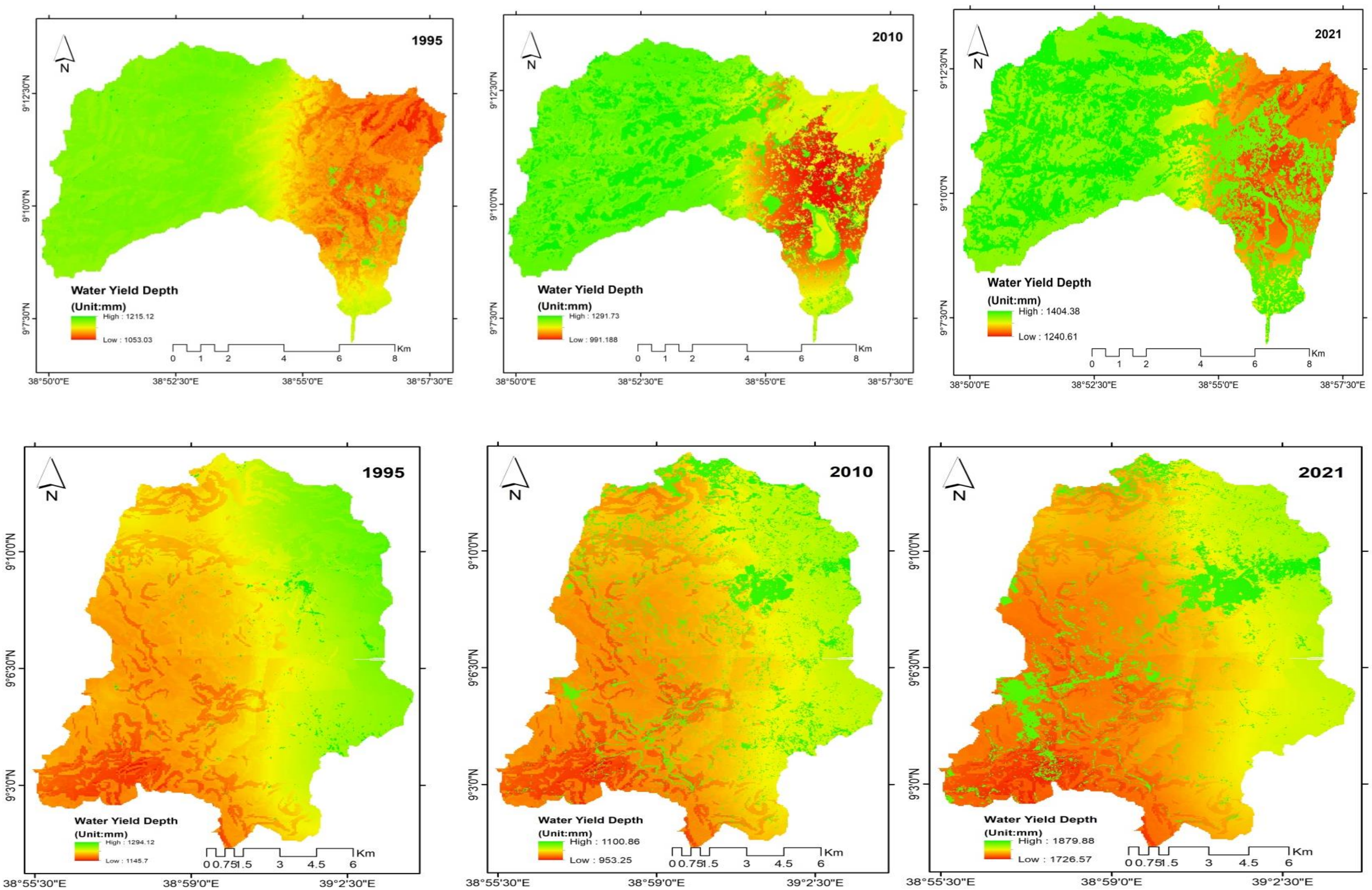

Figure 10: Water yield pattern of Dire and Legedadi watersheds with actual conditions

4.5.2 Influence of Land Use Land Cover Changes and Climate variability on water yield

Climate variability was found to be the major driver of variations in water yield in both watersheds. The effect of climate variability on annual water yield was approximately 99.9% and 73.3%, respectively, while the effect of land use changes was only 0.01% and 26.7% in the Legedadi and Dire watersheds (Table 19; Figure 11, 12). Climate variability resulted in a 32.5% increase in total water yield in the Legedadi and a 14% increase in the Dire between 1995 and 2021 in the absence of LULC change, which was consistent with actual conditions. Although total water yield decreased by 18.9% between 1995 and 2010, a substantial increase of 43.2% was observed in the Legedadi watershed between 2010 and 2021. Similarly, total water yield increased by 6.13% and 8.5%, respectively, between 1995 and 2010 and 2010 and 2021 in the Dire watershed (Table 19; Figure 11, 12). In the absence of climate variability, LULC conversions increased total water yield by 0.72%, 0.68%, and 0.03% in the Legedadi watershed between 1995 and 2021; 1995 and 2010, and 2010 and 2021, respectively; while a 2.7%, 6.5%, and 3.5% reduction was observed in the Dire watershed between 1995 and 2021; 1995 and 2010, and 2010 and 2021, respectively (Table 19; Figure 11, 12).

Table 19: Total and mean water yield under different scenarios

Scenarios	Total Water Yield (10^6 m^3/year) (Water Yield Depth) (mm)		
	1995	2010	2021
	Legedadi		
Actual conditions	259.5(1202)	219.9 (1018.6)	386.6 (1790)
Conditions without Climate variability	259.5 (1202)	261.3 (1210.3)	261.4 (1210.8)
Conditions without Land Use Change	259.5 (1202)	218.2 (1010.5)	384.6 (1781.3)
	Dire		
Actual conditions	111.6 (1149.5)	116.8 (1202)	131.8 (1357.5)
Conditions without Climate variability	111.6 (1149.5)	109.5 (1127.5)	113.5 (1169)
Conditions without Land Use Change	111.6 (1149.5)	118.9 (1224.7)	129.9 (1337.4)

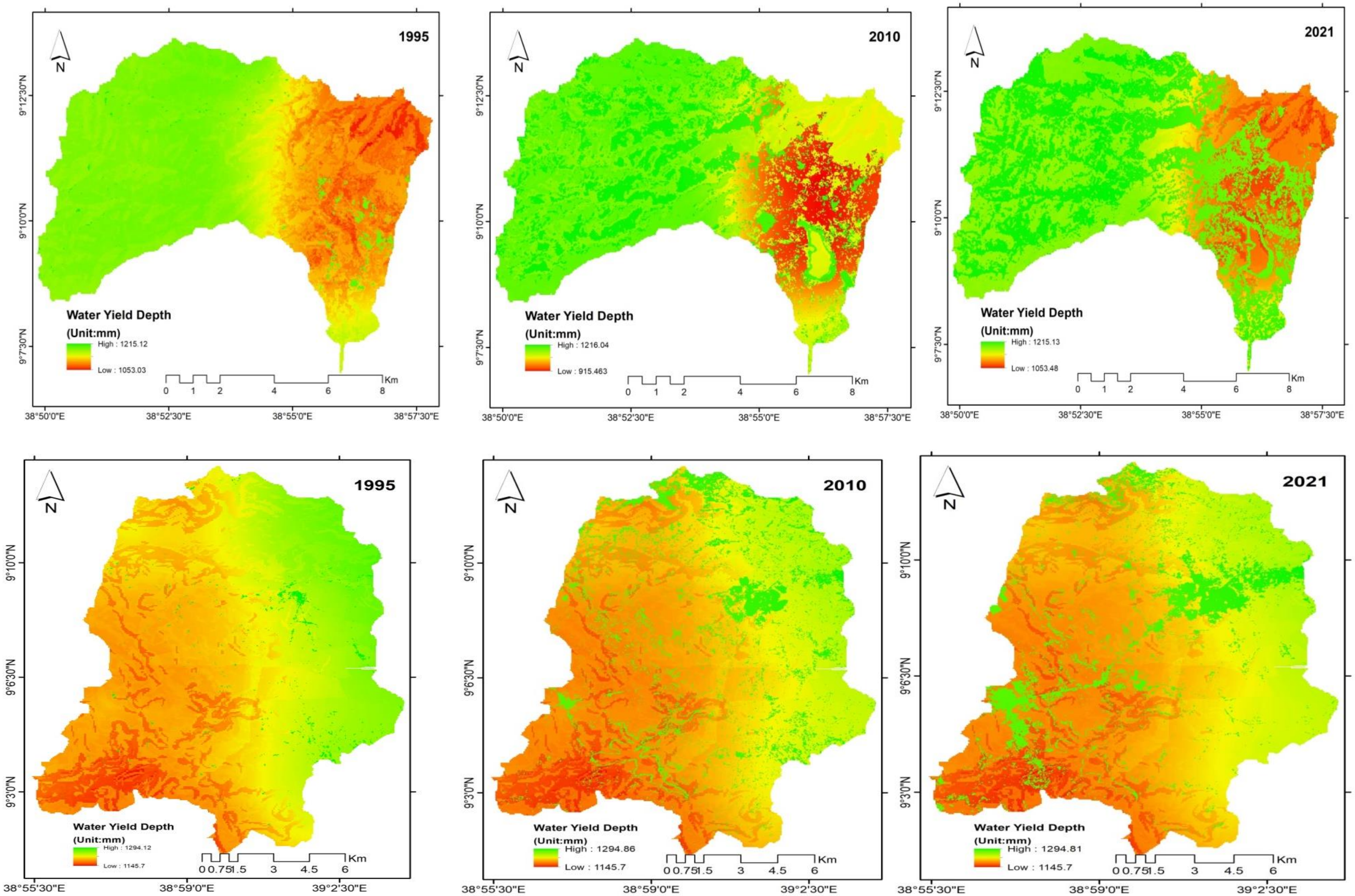

Figure 11: Water yield pattern of Dire and Legedadi watersheds with LU/LC unchanged

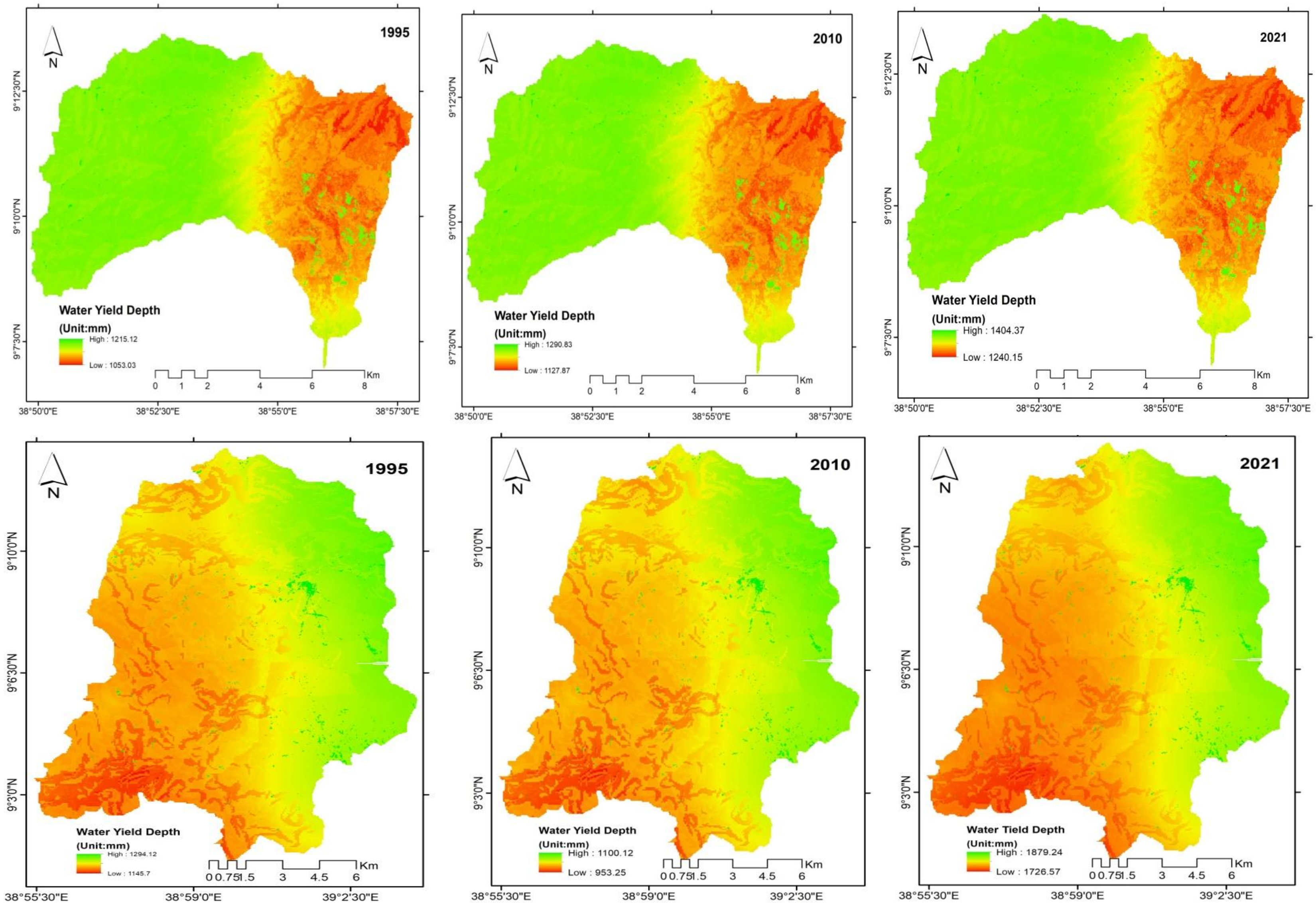

Figure 12: Water yield pattern of Dire and Legedadi watersheds with climate variables unchanged

4.6 Discussions

4.6.1 Annual water Yield

Between 1995 and 2021, the total water yield in the Dire and Legedadi watersheds increased. It has risen from approximately 259.5 MCM in 1995 to approximately 386.6 MCM in 2021 in the Legedadi watersheds. Similarly, the Dire watershed's total water yield has increased from approximately 111.6 MCM in 1995 to approximately 131.8 MCM in 2021. Increased rainfall in the Legedadi watershed from 1302 mm in 1995 to 1887.2 mm in 2021; and 1220 mm in 1995 to 1409.5 mm in 2021 in the Dire watershed could contribute to increased annual water yield in both watersheds. Furthermore, the annual water yield reduction of 18.9% (41.3 MCM) in the Legedadi watershed between 1995 and 2010 could be attributed to a decrease in annual rainfall from 1302 mm in 1995 to 1108 mm in 2010; Mesfin et al., (2019) reported that improved precipitation and lower actual evapotranspiration contributed to higher annual water yields in Ethiopia's Wabe catchment. The two watersheds' average annual surface water potential was estimated to be 86 MCM in the Legedadi catchment and 50 MCM in the Dire catchment (AAWSA, 2011). Our estimates for the Legedadi and Dire watersheds in 2010 are approximately 219.9 MCM and approximately 116.8 MCM, respectively, which are higher than the AAWSA estimates (AAWSA 2011). This could be because the InVEST model does not distinguish between surface, subsurface, and base flow water yield; instead, the model assumes that all water yields from a pixel reach the point of interest and then adds the water yield (Sharp et al., 2020). Human and livestock water consumption is increasing through wells and springs in both watersheds. Deep wells are used in urban areas such as Sendafa, Legedadi, and Dire, whereas shallow wells are used in rural areas (AAWSA, 2011). Thus, the total water abstraction in both watersheds was estimated to be 790 m^3/day, which is less than the required water utilization demand by both livestock and humans in the area (AAWSA, 2011). The current total water yield (518.4 MCM) in both watersheds may be insufficient to meet rising water demands, as rapid land cover changes and human and livestock growth are expected to degrade the quality of ecosystem services throughout the watersheds (Simeneh et al., 2023b). To fully meet the current water demand of Addis Ababa, an estimated supply quantity of 0.8 MCM/day (Van Rooijen and Taddesse, 2009) to 1.2 MCM/day would be required (Tirusew and Fekadu, 2023). This disparity contrasts sharply with current water supply rates from both surface and groundwater, which are estimated to be 0.48 MCM/day, covering only about 40% of the daily water demand (Tirusew and Fekadu, 2023). As a result, a mega water project may be required to meet Addis Ababa's projected 307 MCM unmet water demand by 2030 (Zinabu and Michael, 2020).

4.6.2 Influence of Land Use Land Cover Changes and Climate Variability

Both climatic and LULC dynamics have a significant impact on water yield (Ningrum et al., 2022). Furthermore, there is a strong positive relationship between precipitation and water yield (Pessacg et al., 2015). Similarly, we discovered that the impact of climate variability was greater than that of LULC change in our study. Climate variability had a significant impact on annual water yield in the Legedadi and Dire watersheds (Surafel and Amare, 2022); whereas land use conversion had a minor impact. Climate change has had a greater impact on water yield changes than LULC changes, according to Mo et al., (2021) in the Dongjiang Lake Basin; Lang et al., (2017) in the Sancha River Basin; Pessacg et al., (2015) in the Chubut River Basin; Bai et al., (2019) in Kentucky; and Sangam et al., (2016) in the Bago River Basin. Precipitation is an important variable of climate elements that has a significant impact on water yield (Lang et al., 2017). It occurs primarily naturally because anthropogenic factors have little effect on precipitation. However, both climatic conditions and LULC changes can influence actual evapotranspiration. Although human activities have a significant impact on LULC changes, they have little impact on water yield. The complex process of land conversion could be the main reason for the lower contribution of land use change (Lang et al., 2017). Because various land use conversions resulted in both positive and negative water yields, the overall effect of land use changes on water yield was not explicitly illustrated. The effect of LULC change on water yield was slight (0.01%) in the Legedadi watershed, but it had a substantial influence on annual water yield (26.7%) in the Dire watershed. This could be due to the Dire watershed's conversion of grassland, which covered approximately 48.5% of the total land cover in 1995, to bare land, cultivated land, and reservoir in 2021. Changes in LULC affect soil conditions, erosion, biodiversity, and the underlying surface, all of which increase runoff. This process in the watershed will affect reservoir capacity and, eventually, water yield (Lang et al., 2017). Conversion of land use is a complicated process (Nie et al., 2011). The forest's deep root structure allows the plant to extract deep soil moisture, and the increased ability to intercept rainfall may result in lower water yields (Lang et al., 2017). As a result, converting cultivated or grassland to forest land may result in a lower water yield (Dagnachew et al., 2003; Sisay, 2014; Lang et al., 2017; Awraris Mamo, 2017). Cultivated and grassland, in contrast to forest land, have shallow root systems and lower interception capability, which may result in increased water yield. Thus, the growth of shrubs and grass would be more appropriate to improve water yield capacity (Guodong et al., 2020), especially in areas where fresh water is scarce. Furthermore, in urban areas, the formation of a rainproof layer of surface covers such as concrete, asphalt, and cement could contribute to rapid runoff, potentially reducing soil moisture and increasing water yield. Thus, the conversion of natural vegetation to urban areas has resulted in an increase in water yield in the area, but it may be detrimental to long-term water provision in the study watersheds. As a result, watershed restoration inventions must consider the spatial patterns and ecology of landscapes.

4.7 Conclusion

The InVEST model is critical in determining water yield (Lian et al., 2020). Thus, this study quantified annual water yield and then compared the effects of land use and climate variability on water yield in the Dire and Legedadi watersheds using the InVEST water yield model. Watersheds are critical to the overall socioeconomic development of Addis Ababa and its environs. Although water yield and depth increased in both watersheds between 1995 and 2021 due to increased precipitation, a decrease in precipitation resulted in a lower water yield and depth between 1995 and 2010, particularly in the Legedadi watershed. Changes in land use have had a less impact on water yield in both watersheds, but the expansion of urban and cultivated land may be susceptible to water yield. The findings of this study can be used to guide watershed management interventions (Wang et al., 2016) that include further investigation of landscape patterns to optimize and maintain the ecosystem's functionality of pristine ecological processes in watersheds.

Chapter Five

5 Analyzing Outdoor Recreational Potential in the Dire and Legedadi watersheds of central Ethiopia.

5.1 Abstract

Outdoor activity is increasingly being recognized as a common practice in Addis Ababa and its surrounding landscapes. Many operators are utilizing social media platforms to promote recreational products. The landscapes surrounding the city of Addis Ababa are becoming the primary destination for hiking. Using the ArcGIS tool, this study evaluated the potential for outdoor recreation in the Dire and Legedadi landscapes by employing previously established indicators. The findings revealed that the highest weight was assigned to the measure of natural intactness (1.39), followed by landscape greenness (1.35), and topographic variability (1.16). Conversely, the presence of water features, steepness of the terrains, and level of disturbance received the lowest weights, with values of 1.03, 0.83, and 0.42, respectively. These landscapes possess considerable potential for outdoor recreation, with approximately 19% and 23% of the Legedadi and Dire landscapes, respectively, exhibiting supreme recreational potential. Furthermore, the existence of diverse bird species further enhances the significance of the area for the development of ecotourism and biodiversity conservation. Therefore, designating this area as a key biodiversity area would contribute to the effective implementation of global biodiversity frameworks.

Keywords: ArcGIS, Outdoor Recreation, Potential, Watershed, Ethiopia

5.2 Introduction

Nature-based recreational activities are among the most relevant ecosystem services provided by landscapes (Gómez and Barton, 2013; Elmqvist et al., 2016) and can be considered as one of the limited natural recreational services for human beings. In urban ecosystems, nature-based recreational activities encompass a wide range of human-dominated green infrastructure (Takano et al., 2002; Camps et al., 2015) as well as semi-natural and natural areas. The proximity of natural and semi-natural areas to metropolitan cities has become crucial for outdoor recreation (Chiara et al., 2018). Exposure to nature, particularly through outdoor recreation, is widely believed to enhance human well-being (Sandifer et al., 2015) by fostering aesthetic experiences (Church et al., 2014), improving mental health (Bratman et al., 2019), and promoting social cohesion (Hernández-Morcillo et al., 2013).

According to Sharp et al. 2020, outdoor recreational activities heavily rely on ecological factors, including species richness (Loureiro et al., 2012), habitat diversity (Neuvonen et al., 2010; Loureiro et al., 2012), precipitation (Loomis and Richardson, 2006), and temperature (Richardson and Loomis, 2005), as well as other attributes such as infrastructure and cultural attractions (Mills and Westover, 1987; Hill and Courtney, 2006). Other types of green infrastructure found in and around cities consist of urban and peri-urban forests (Bertram and Larondelle, 2016), tree-lined streets (Takano et al., 2002), peri-urban agriculture (Zasada, 2011), brownfields and vegetable gardens (Pueffel et al., 2018), and abandoned areas (Foster, 2014), all of which support various nature-based recreational activities and experiences (Caspersen and Olafsson, 2010).

Agricultural landscapes play a vital role in promoting tourism and enhancing the quality of life in rural areas (Kati et al., 2018). The Dire and Legedadi landscapes primarily consist of agricultural lands, serving as the main source of income. The rural population in both watersheds heavily relies on a mixed crop-livestock system (Simeneh et al., 2023a). Crop production and livestock farming contribute significantly to the income of small-scale farmers (Simeneh et al., 2023a). However, the expansion of cropland could negatively affect the recreational services.

The aesthetic appeal of landscapes and the amenities associated with the natural environment contribute to the enjoyment of outdoor recreation. Activities such as trekking, hiking, boating, and bird watching require specific landscape features, including diverse topography, suitable habitats, and the preservation of the area's integrity (Nigussie et al., 2021). The Ethiopian government has acknowledged tourism as a critical sector for development, leading to the implementation of significant projects aimed at promoting tourism destinations and improving infrastructure across the country. Despite the rapid growth of outdoor activities in the areas surrounding Addis Ababa, no empirical research has been conducted to evaluate the recreational potential of protected and unprotected landscapes.

The assessment of the potential for recreational activities is of great significance to effectively strategize and formulate plans for outdoor recreational endeavors (Burkhard et al., 2012; Castillo-Eguskitza et al., 2018). To measure the magnitude of recreational services, it is possible to employ a comprehensive approach to ecosystem services that utilizes comparable biophysical metrics (Tratalos et al., 2016; Liu et al., 2020). The concept of recreational potential can be defined as the extent of available recreational space within a given landscape (Liu et al., 2020; Tallis et al., 2012). To achieve this, previously established predictors such as the amount of greenery in the area, the presence of water features, the level of naturalness, the configuration of the landscape, the degree of disturbance (including proximity to residential areas), and the variety of recreational activities were utilized (Nigussie et al., 2021). Furthermore, the characteristics of the study landscape were also considered. The application of the ArcGIS tool has been tested in various recreational potential assessments (Snyder et al., 2008; Silberman and Rees, 2010; Sherrouse et al., 2011; Selemawi et al., 2021; Jeetendra and Raj, 2023). Therefore, the objective of this assessment was to map and quantify the recreational potential of the landscape using the aggregated ArcGIS software, to utilize this information for appropriate development of outdoor recreational activities in the watersheds.

5.3 Materials and Methods

5.3.1 Mapping Recreational Potential

The methodological technique was developed based on a comprehensive examination of previous approaches employed for the mapping of cultural environmental services (Frank et al., 2013; Nahuelhual et al., 2013; Plieninger et al., 2013; Casado-Arzuaga et al., 2014; Paracchini et al., 2014; van Berkel and Verburg, 2014; Lorena et al., 2015; Schägner et al., 2018; Ghorbanzadeh et al., 2019; Selemawi et al., 2021; Jeetendra and Raj, 2023). To assess and delineate the most suitable areas within landscapes for recreational activities, a straightforward weighted overlay analysis was executed (Nino et al., 2017; Edgars and Zane, 2020). The study made use of previously established indicators for recreational purposes (Nigussie et al., 2021). To quantify and map the recreational potential of the landscapes, the biophysical environment of the watersheds, encompassing habitat quality, Normalized Difference Vegetation Index (NDVI), Digital Elevation Model (DEM), Slope (derived from the DEM), settlement (derived from the LULC), and water reservoirs, was employed as a proxy (Table 20; Figure 13; 14). The NDVI serves as a measure of greenness, which can be summarized and applied across various areas of interest for comparison (Tucker, 1979; Vukomanovic and Orr, 2014). The state of biodiversity can serve as a foundational proxy tool for evaluating the quality of the landscape ecosystem (Havlicek and Mitchell, 2014). Consequently, habitat quality was employed as an indicator of the intactness of the natural environment (Polasky et al., 2011; Villamagna et al., 2013).

Table 20: Indicators and their proxy variables used for quantifying and mapping recreational potential.

Indicators	Proxy predictors	Data source
Greenness	NDVI	Image analysis performed in ArcGIS 10.5 tool
Intactness	Habitat quality	Habitat quality assessment using InVEST HQ model Simeneh *et al.*, 2023b
Water features	Reservoir and rivers	The land classification made by Simeneh *et al.*, 2023a and extraction from DEM
Mosaic landscape	DEM and Slope	Spatial analysis tool using ArcGIS 10.5
Disturbance level	LULC	The land classification made by Simeneh *et al.*, 2023a

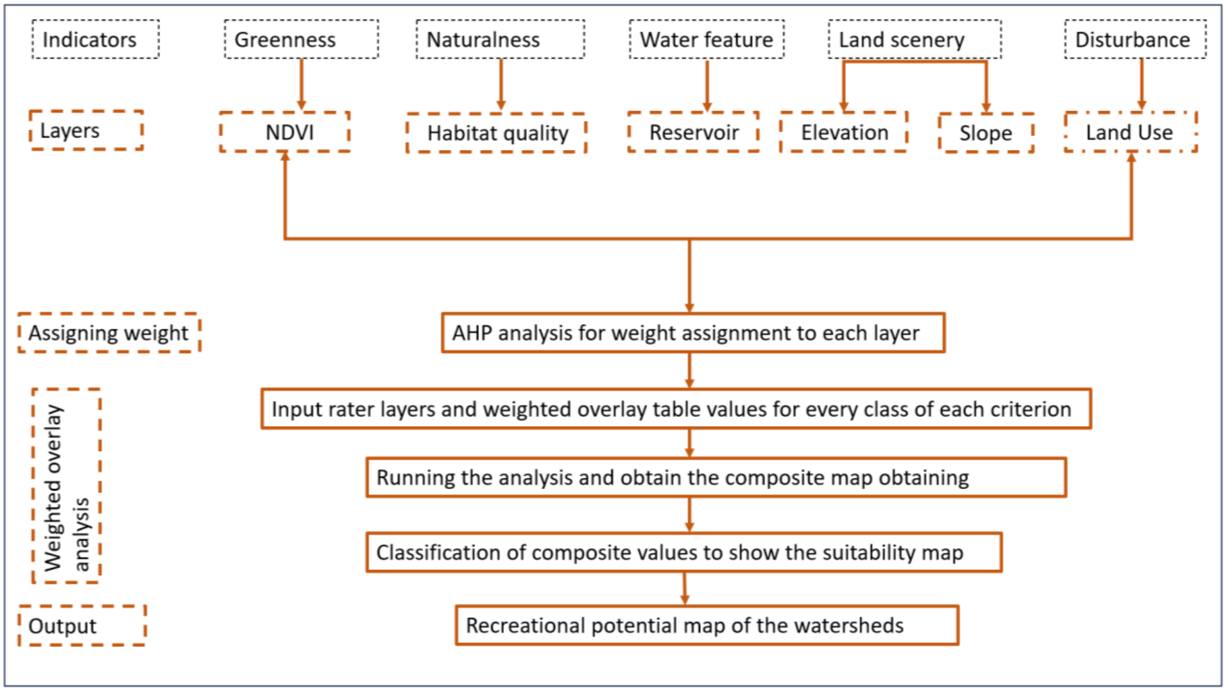

Figure 13: Summary of methods used to assess recreational potential in landscapes.

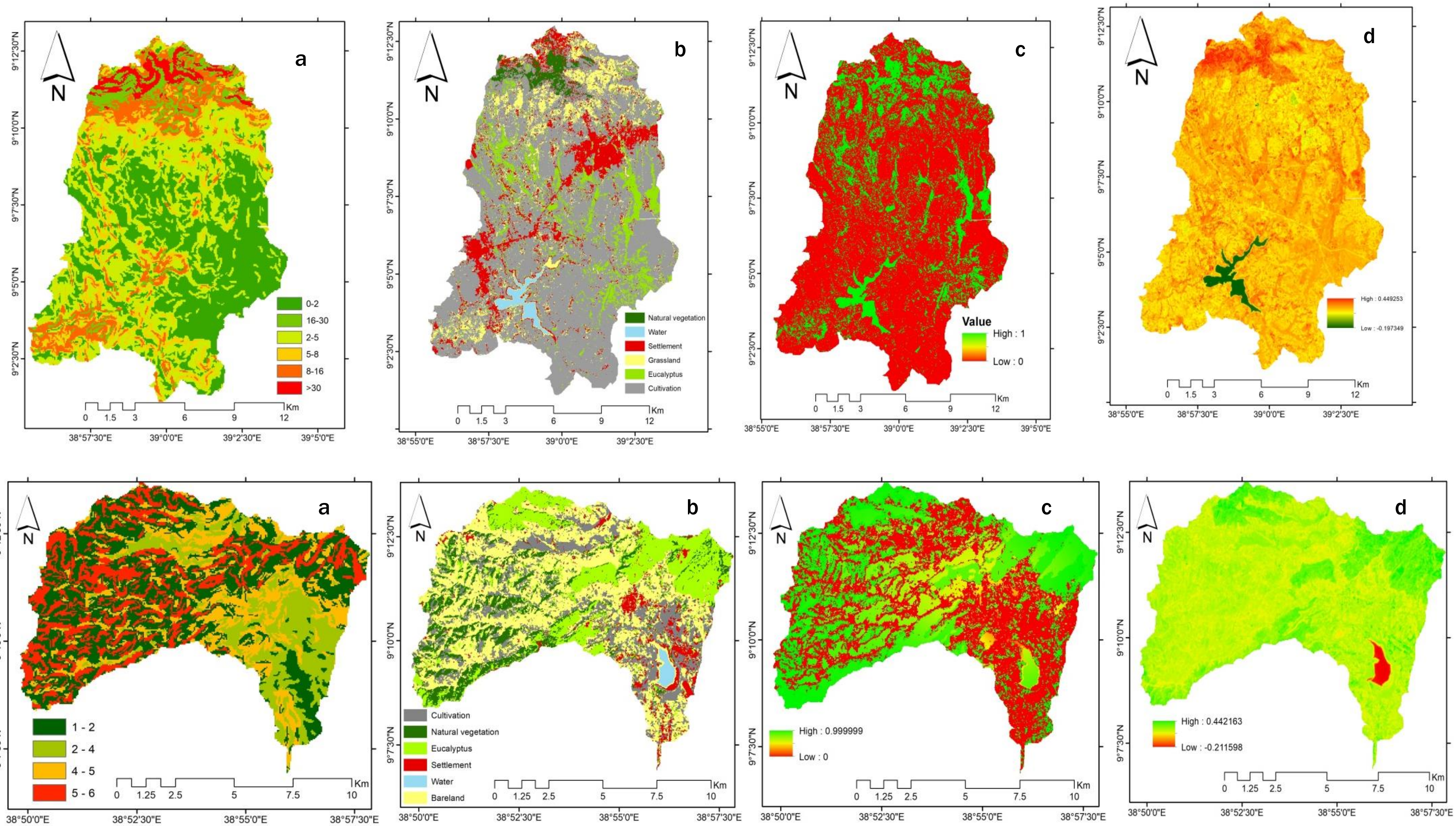

Figure 14: The biophysical environment of the watersheds, includes layers of Slope (a), LULC (b), HQ (c), and NDVI (d) used to quantify and map recreational potential.

5.3.3 The Application of AHP and Predictors Weight Determination

The evaluation of recreational services can be conducted using the ArcGIS tool and the Analytical Hierarchy Process (AHP) method, as suggested by Nahuelhual et al. (2013). AHP, a widely employed Multi-Criteria Decision Making (MCDM) technique in spatial multi-criteria decision analysis (Kordi and Brandt, 2012), provides a structural foundation for quantifying the evaluation of decision factors and criteria through a pair-wise method (Arabinda, 2003). Consequently, the relative importance of spatial criteria weighting is determined, thus measuring the significance of each spatial criterion. Subsequently, a pairwise assessment to estimate the relative importance of attributes and construct an assessment matrix of the ranks for every hierarchical stage was made (Saaty and Vargas, 2000; Gomal et al., 2011; Nahuelhual et al., 2013) (Table 21). The final recreational maps are then classified into 4 classes (High, Medium, Low, and Poor) using natural breaks (Casado-Arzuaga et al., 2014; Lorena et al., 2015; Selemawi et al., 2021).

Table 21: Pairwise comparison of the nine-point rating scale after Saaty and Vargas (2000)

Importance	**Definition**	**Explanation**
1	Equal significance	Two criteria contribute equally to the objective criteria
3	Moderate significance	Judgments and experience slightly favor one criterion over another
5	Strong significance	Strongly favored by judgments and experience
7	Very strong significance	A criterion is strongly favored, and its dominance established in practice
9	Extreme importance	The evidence favoring one criterion over another is of the highest probable order of affirmation
2,4,6,8,	Intermediate values	When adjustment is needed
Reciprocals	Values for inverse comparison	

The individual recreational potential maps of the selected criteria had been weighted using their importance (Table 22). The total rating for each alternative was calculated following Zardari et al., 2015 and Pramanik's 2016 method by multiplying the significance of weight assigned to every characteristic by the scaled value given for that attribute to the alternative and then summing the product's overall attributes (Cengiz and Akbulak 2009; Pramanik 2016; Pande et al., 2021a). Thus, all the layers had been overlaid by recognizing cellular values to the same scale, giving a weight value to the individual criterion, and integrating the weight cell values as given below:

$$LS = \sum_{i=1}^{n} Wi\, Xi \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{(14)}$$

Where LS= total recreational potential score, Wi = the weight assigned to each selected recreational potential criteria; Xi = the factor score (cells) of i recreational potential criteria, and n = the total number of land capability criteria (Simeneh et al., 2022).

The consistency of the judgment matrix is examined with the calculation of the consistency index (CI) as defined by the equation (Mansouri, 2014; Simeneh *et al*., 2022):

$$\mathrm{CI} = \frac{(\lambda\mathrm{max} - \mathrm{n})}{(n-1)} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \textbf{(15)}$$

Where CI = consistency index, λ_{max} = matrix's most significant or principal eigenvalue that could be calculated from the matrix, and n = order of the matrix (Ying et al., 2007; Zardari et al., 2015; Pramanik, 2016).

Also, in the AHP, a measure of consistency ratio (CR) was calculated to indicate the randomized probability of matrix judgments as follows:

$$CR = \frac{CI}{RI} \text{..............................} \textbf{(16)}$$

Where CI = average of the resulting consistency index, depending on the order of the matrix given by Malczewski (1999; 2000), and RI (Random Index) = values for matrices of different sizes (Saaty, 2003).

Table 22: Criteria classification for recreational potential

Indicators category	Predicators	Eigenvector weight	Significance (%)
Greenness	NDVI	1.35	23.08
Naturalness	Habitat quality	1.39	23.08
Water features	Reservoirs	1.03	11.54
Scenery	Elevation	1.16	23.08
	Slope	0.83	11.54
Disturbance	LULC	0.42	7.69
Consistency Index (CI) = -1.04 and Consistency Ratio (CR) = -0.93			

5.4 Results and Discussion

The individual recreational potential maps of the selected criteria had been weighted using their importance. The rating of the six evaluation criteria revealed the importance of every criterion in the order of intactness, greenness, topography, water feature, steepness, and recreational disturbance (Table 23). Intactness obtained the highest weight (1.39) followed by Greeness (1.35), and topography (1.16) respectively. The water feature, steepness, and recreational disturbance obtained the bottom 1.03, 0.83, and 0.42 respectively (Table 23).

A total of 312 km^2 of land was computed to determine the landscapes' recreational potential. Out of this about 40.53 km^2 (18.82%) and 22.95 km^2 (24.13%) of the land have the highest recreational potential and approximately 21.17 km^2 (9.83%) and 17.7 km^2 (18.61%) have moderate recreational potential in the Legedadi and Dire landscapes respectively. Whereas approximately 143.7 km^2 (66.7%) and 35.6 km^2 (37.4%) have the poorest recreational potential and about 10 km^2 (40.64%) and 18.91 km^2 (19.88%) of the landscapes have the lowest recreational potential in the Legedadi and Dire landscapes, respectively (Table 24; Figure 15). The environmental elements are related to the complexity and connectivity of landscapes that regularly have a strong tremendous relation to the biodiversity richness on various scales (Amici et al., 2015). Consequently, the recreational potential in this study is largely associated with the landscape's environmental factors. Therefore, the landscape's excessive recreational potential area can be essential for landscape conservation and recreational activities. The rugged terrains having steep slopes and gorges have higher potential; accordingly, these lands should have extra multifunctional surroundings services advantages.

The poorest and lowest recreational potential areas of the landscapes account for more than 70% while moderate and highest recreational potential areas account for below 30% within the Legedadi watershed. In contrast, the poorest and lowest recreational potential areas account for about 57% at the same time while the highest and slightest potential regions account for approximately 43% in the Dire watershed. In each landscape, the poorest and lowest recreational potential areas are mainly located in the area in which congested settlement, bare land, and agricultural activities take place; while moderate and highest recreational potential areas are found in an area in which natural vegetation is ample and land adjacent to the reservoir.

The Dire has a higher annual cultural service value of US$ 4,843 than the Legedadi US$ 1491 (Simeneh et al., 2023a). Most of the recreational potential regions are in Mount Bereh and surrounding areas. Recently, mount Bereh and its surroundings have become a favorite place for paragliding activities as the rugged terrain and appropriate weather circumstance favors a smooth adventure flying activity. Therefore, the development of new tourism products would enhance tourism in the landscapes. Furthermore, the Dire and Legedadi water reservoirs can be among the important bird conservation areas, due to their significance for bird conservation mainly for the Afrotropical biome birds and large aggregation of waterbirds. charismatic bird species have been observed in the study areas. The richness and abundance of waterbirds also attract people to the area by providing them with a unique experience of witnessing the birds in their natural habitat. The presence of waterbirds further suggests that the area could be attractive for conservation activities for the protection of species and the ecosystem and could also make the area important for the development of ecotourism. An increasing number of bird species can provide higher satisfaction to visitors (Hepburn et al., 2021) bring additional economic resources to local communities, and contribute to biodiversity conservation (Schwoerer et al., 2022). However, recreation can also lead to displacement, influence breeding success, and reduce the survival *of* sensitive bird species (Rodríguez-Prieto et al., 2014).

Table 23: Description of the input layers used in weighted overlay analysis.

Criteria	Land suitability classes				Weight	Influence (%)
	S1 Highly suitable	S2 Moderately suitable	S3 Marginally suitable	S4 Very low		
Elevation	>3000	2700-3000	2500-2700	<2500	1.16	23.08
Slope	0-5°	5-20°	20-50°	>50°	0.83	11.54
Greenness (NDVI)	>0.5	0.5-0	0-(-0.5)	<-0.5	1.35	23.08
Naturalness (HQ)	>0.75	0.5-0.75	0.25-0.5	<0.25	1.39	23.08
Disturbance (LULC)	Natural vegetation, Reservoirs, Grassland	Plantation	Cultivation	Settlement	0.42	7.69
Proximity to water reservoirs	0-500 m	500-1000 m	1000-2000 m	>2000 m	1.03	11.54

Table 24: Quantified recreational potential in the study landscapes

Classes	Quality	Watersheds			
		Legedadi		Dire	
		Area (km^2)	Percent	Area (km^2)	Percent
S1	High	40.53	18.82	22.95	24.13
S2	Moderate	21.17	9.83	17.7	18.61
S3	Low	10	4.64	18.91	19.88
N	Poor	143.7	66.71	35.56	37.38
Total		215.4	100	95.12	

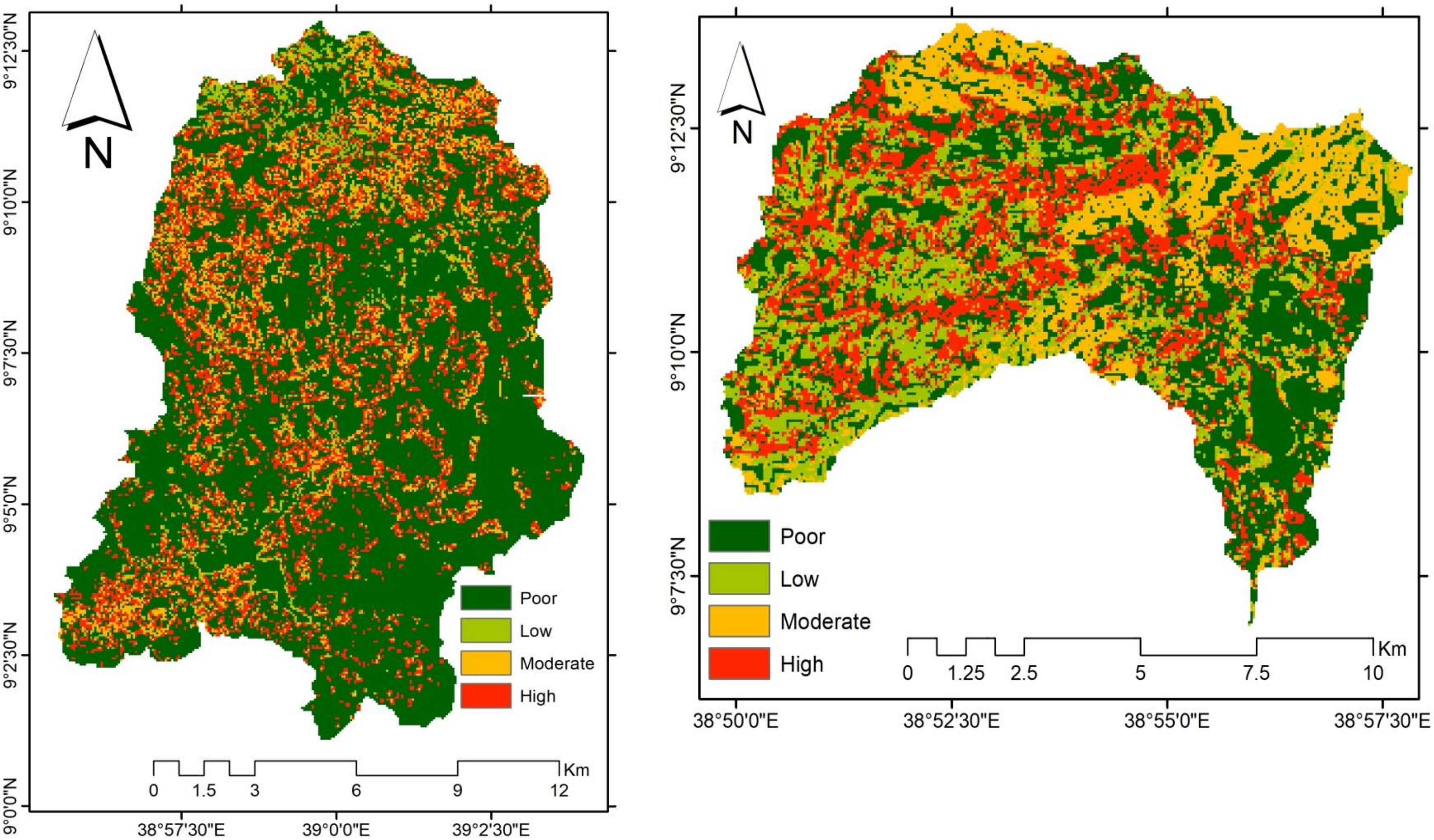

Figure 15: Images depicting the spatial recreational potential map of the study landscapes.

5.6 Conclusions

Outdoor recreation is important for human well-being. It also helps local communities to earn money and improve social interaction. The study's findings demonstrated that landscapes have tremendous recreational potential. High recreational value areas can be used to develop ecotourism and are also important for biodiversity conservation. Because there is a growing interest in outdoor recreational activities in the area, proper ecotourism planning can benefit the local community. Furthermore, the presence of such a diverse bird species may make the area valuable enough to be designated as an important bird area, which could support effective biodiversity conservation and hasten the implementation of a global biodiversity framework. Finally based on the results of this study ecotourism planning and management interventions were proposed to improve the recreational value of the landscapes.

Chapter Six

6 Analysis of Land Suitability for Apple-based Agroforestry Farming in Dire and Legedadi watersheds of Ethiopia.

6.1 Abstract

This study was conducted to evaluate land suitability for apple farming in the Dire and Legedadi watersheds of the central highlands of Ethiopia. Attributes that determine apple growth were categorized into environmental, soil, climate, and land management factors. The land evaluation methodology developed by FAO (1976) was applied in six steps. First, nine thematic layers are prepared. Second, pair-wise comparison matrices were performed using AHP. Third, thematic layers are reclassified. Fourth, weights are assigned to each class. Fifth, weighted overlay analysis was performed to produce a land suitability map. Finally, the land suitability map was classified into high, moderate, marginally suitable, and unsuitable categories. Soil type received the highest weight of 1.98 followed by elevation and LULC of 1.51 each. The mean temperature, rainfall, soil pH, and soil drainage weight were 1.41, 0.94, 0.56, and 0.52 respectively. Whereas the slope and aspect weighted the lowest at 0.38 and 0.19 respectively. Out of the total area of the watersheds, about 14 km^2 (6.7%) and 12.34 km^2 (13.1%) are highly suitable for apple farming in the Legedadi and Dire watersheds respectively. Whereas, about 113.35 km^2 (53.8%) and 42.54 km^2 (45.2%) of land are not suitable in the Legedadi and Dire watersheds respectively. Landholders who play a pivotal role should be incentivized to grow perennial crops (e.g., apple trees) to enhance environmental income and alleviate poverty.

Keywords: Agroforestry, Ecosystem services, Apple, Land suitability, Watersheds

6.2 Introduction

Agroforestry provides numerous environmental and socioeconomic benefits that have larger local and regional importance (Jose, 2009). It is a highly recognized tool for integrated and sustainable land use schemes that enhance ecological and economic viability (Mbow, 2015; Pande et al., 2021). Agroforestry has also directly influenced agricultural productivity through the reduction of soil erosion, maintaining moisture and organic matter of soil, controlling pests, enhancing pollination, and increasing resilience to climate impacts (Hooper *et al.,* 2005; Mbow, 2015; Roointan et al., 2018; Do *et al.,* 2020). Moreover, agroforestry systems have a multifunctional ecosystem role, which provides multiple ecosystem services; including the provision of food and fiber; regulating climate, water, and soil; enhancing water and air quality; improving nutrient recycling which increases soil and maintains fertility; critical species for ecological functioning and aesthetic values are among the principal ecosystem services of agroforestry (McAdam *et al.,* 2009; Torralba *et al.,* 2016; Pande et al., 2021).

The application of an agroforestry system plays an important role in the conservation of natural resources, especially soil and water conservation (Pande and Moharir, 2015; Rajesh et al., 2021;). Despite its principal component of soil and water conservation, integrated watershed management is also critical to maintaining biodiversity resources and enhancing crop production and livestock development (Abhijit et al., 2013). Thus, it can be considered an invaluable tool for watershed management in which it backstops sustainable use of natural resources and enhances the well-being of people. The appropriate application of agroforestry practice can increase the productivity of the agricultural landscape which has a greater implication for ecological functioning and biodiversity conservation (Rohit *et al.,* 2019). Moreover, agroforestry provides a unique opportunity to combine the twin objectives of climate change adaptation and mitigation.

It has been increasingly recognized as a useful tool to mitigate climate change impacts (Verchot *et al.*, 2007; Shahid et al., 2021) and provides an unparalleled contribution to enhancing ecosystem resilience.

Apple is a deciduous fruit tree that grows in temperate climate zones where most commercial varieties fulfill their required chilling temperature. Although apple trees originated and their environmental requirement is appropriate in temperate regions, many tropical and subtropical countries are adopting the tree (Williams and Tzoc, 1990). The lower temperatures at much higher altitudes allow the fruit to reach of its chilling requirement (Osborne, 2000). The applications of apple-based agroforestry systems have significant contributions to biomass production and mitigate climate change impacts (Zahoor *et al.,* 2021; Pande et al.,2021).

In Ethiopia, the apple tree was introduced six decades ago by European missionaries (Bezabih, 2009; Ashebir *et al.* 2010; Girmay *et al.,* 2014). Later, the agricultural-led economic growth strategy provided massive emphasis on the introduction of temperate fruits to different parts of the country (Handoro and Gemu, 2007). Most farmers typically produce apples to supplement their meager household income (Zerihun *et al.,* 2019). Hayesso (2008) and Girmay *et al.,* (2014) reported that the annual apple fruits production is about 50 metric tons in Ethiopia which is below the demand; consequently, Ethiopia imports about 350 metric tons of apple fruits annually (Hayesso, 2008; EHDA, 2012). This indicates the presence of a promising market for apple growers in Ethiopia.

The rapidly growing populations coupled with environmental and social changes such as devastating habitat destruction and degradation for cultivation practice and expansion of urbanization are adversely affecting the characteristics of the upper headwaters of Addis Ababa the Dire and Legedadi watersheds (AAWSA, 2011; 2016; Gebreegziabher *et al.,* 2021). Consequently, the widespread alteration and fragmentation of the natural land cover became the greatest threat to the ecosystems. The application of a properly designed apple-based agroforestry system in both watersheds will largely contribute to the improvement of ecosystem services (e.g., water quality and quantity through reducing siltation) in the watersheds.

More recently, several land evaluation studies were made that assessed land suitability for various crops in Ethiopia including Agidew, (2015) for sorghum and barley crops; Girma *et al.*, (2015) for maize, wheat, and sorghum; Hailu *et al.*, (2015) for barley and wheat; Nahusenay and Kibebew (2015) for barley, wheat, bean, and lentil; Liambila and Kibret (2016) for sorghum, maize, bread wheat, sweet potato, and soybeans; Motuma *et al.,* (2016) for wheat and sorghum and; Hamere and Teshome, (2018) for wheat and barley. More recently, Getachew *et al.*, (2022) have assessed land suitability for apple (*Malus domestica*) production in the Sentele watershed of southern Ethiopia. Further, several studies have demonstrated the socioeconomic benefits of apple farming in Ethiopia including Handoro and Gemu, 2007; Hayesso, 2008; Bezabih, 2009; Ashebir *et al.* 2010; Lemlem *et al.*, 2013; Fetena *et al.*, 2014; Girmay *et al.*, 2014; Girmay *et al.*, 2014; Sintayehu *et al.,* 2017; Behailu and Kebede, 2018; Tamirat and Muluken, 2018; Zerihun *et al.*, 2019; Tesfaye and Amene, 2022. The Addis Ababa Water and Sewage Authority (AAWSA) has grown apple trees along the buffer areas of water reservoirs. However, this was practiced without land suitability evaluation. Therefore, this study aims to assess the land suitability to provide the first comprehensive information for apple farming in the study area for robust intervention to enhance the ecosystem service of the watersheds.

6.3 Methods

6.3.1 Methodological approach

The methodological approach for the land evaluation procedure was designed based on the Food and Agriculture Organization (FAO) land suitability guideline (FAO, 1976; 2007b) and a review of similar works such as the study made by Karma and Gibji (2021) and Getachew *et al.*, (2022) land evaluation assessment for apple tree (Figure 16). Land suitability is a multi-criteria analysis that involves several parameters; therefore, the evaluation process requires a multi-criteria approach (Rabia and Terribile, 2013). Accordingly, based on expert opinion from Holeta Agricultural Research Centre (HARC), Addis Ababa Water and Sewage Authority (AAWSA), and field observation and review of literature; apple tree was chosen for land suitability evaluation in the watersheds as the practice will enhance ecosystem services and income for the local economy.

6.3.2 Factors Affecting Apple Growth

The identified influential factors were categorized into the soil, topographic, land management, and climate data (Kim and Shim, 2018; Karma and Gibji, 2021; Getachew *et al.*, 2022) for the assessment of land suitability evaluation. A total of nine factors that influence apple growth were selected and prioritized. These are soil type, soil drainage, soil pH, rainfall, current land use, land cover, mean temperature, elevation, slope, and aspect (Table 25; Figure 16).

6.3.3 Creation of Thematic Maps

All these files were extracted for the study watersheds before running the spatial analysis model in ArcGIS 10.5. The input raster’s were reclassified into different classes. Once all the criteria were recalculated, the "Weighted Overlay" tool was used to produce the suitability classes. For each criterion, a weight factor was assigned when producing the result. Finally, the Weighted Overlay Analysis (WOA), an effective technique to resolve spatial complexity in suitability analysis was applied to generate the land suitability map (Girvan et al. 2003; Pramanik 2016; Taddese 2014). The final suitability map was reclassified based on FAO (1976): these are highly suitable, moderately suitable, marginally suitable, and not suitable. Highly suitable areas show the most favorable/the best biophysical conditions for apple growth in the study area. The moderately and marginally suitable areas indicate the next priority areas which need scrutiny of factors and decisions on the feasibility of investment over other opportunities. The unsuitable area represents those sites constrained by reservoirs, settlements, and other land use types (Table 26).

Table 25: Indicators and their proxy variables used for assessing land suitability for apple farming in the study watersheds.

Parameters	Indicators	Proxy layers	Data source
Topography	Slope	Slope	Extracted from DEM using spatial analysis tool in ArcGIS 10.5
	Elevation	Elevation	
	Aspect	Aspect	
Soil	Soil drainage	Soil drainage	Obtained from Oromia Water Works Design, Construction and Supervision Enterprise (OWWDCSE)
	Soil type	Soil type	
	pH	pH	
Climate	Rainfall	Rainfall	Obtained from NMA then interpolation algorithm performed using spatial analysis tool in ArcGIS 10.5
	Mean temperature	Temperature	Obtained from NMA then interpolation algorithm performed using spatial analysis tool in ArcGIS 10.5

Land management	Land use	LULC	Land classifications and analysis were made in the ArcGIS 10.5 tool

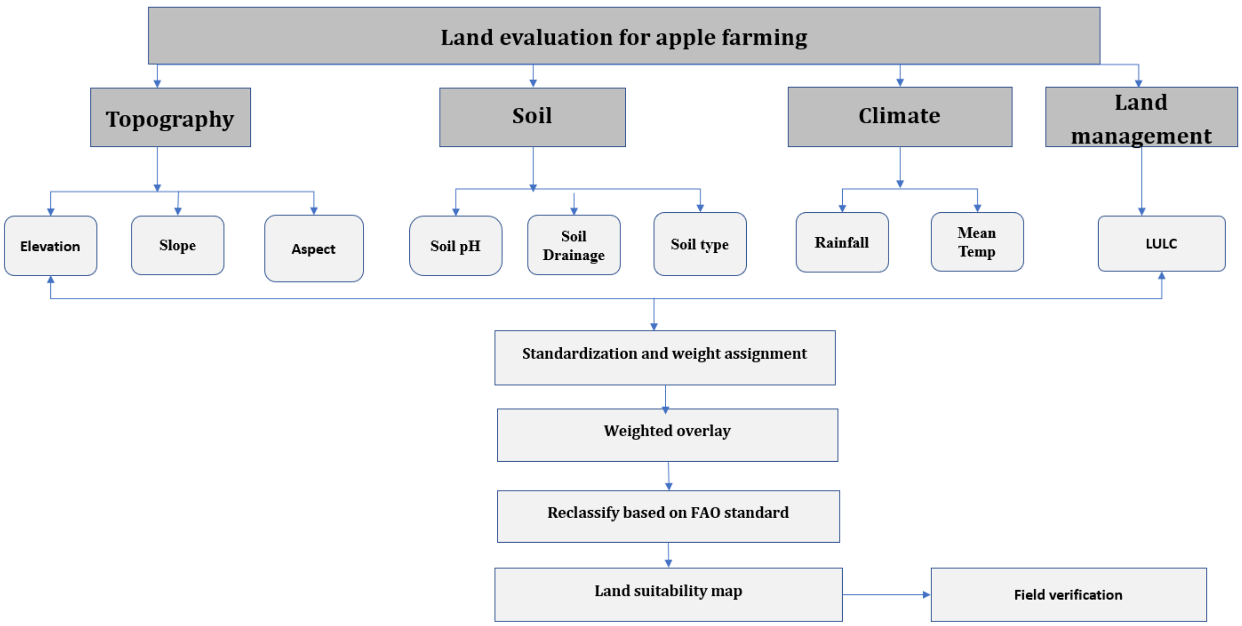

Figure 16: Summary of methods applied to evaluate land suitability for apple-based agroforestry farming in the watersheds.

Table 26: Land suitability classification, descriptions, and numerical scale (FAO, 1976)

Classes	Suitability	Description	Scale
S1	Highly suitable	Land without significant limitations. This land is not perfect but is the best that can be obtained.	1
S2	Moderately suitable	Land that is suitable but has limitations that either reduce productivity or increase the inputs needed to sustain productivity compared with those	2

		needed on S1 land.	
S3	Marginally suitable	Land with limitations so severe that benefits are reduced, and/or the inputs needed to sustain production are increased so that this cost is only marginally justified.	3
N	Not suitable	Land that cannot support land use on a sustained basis, or land on which benefits do not justify necessary inputs.	4

6.3.4 The application of AHP and predictors weight determination

The integration of the AHP method with ArcGIS provides a powerful and accurate analysis of land suitability for agroforestry (Chuma *et al.*, 2021). The AHP method provides a structural basis for quantifying the comparison of decision elements and criteria in a pairwise technique (Arabinda, 2003). The method evaluates the relative significance of all parameters by assigning weight to each of them in hierarchical order to ensure the credibility of the relative significance. AHP also provides measures to determine the inconsistency of judgments mathematically (Saaty, 1980). The AHP employs an underlying nine-point recording scale to rate the relative preference on a one-to-one basis of each factor (Saaty, 1980; Saaty and Vargas, 2000). This nine-point scale used in analytical hierarchy studies ranges from 1 (indifference or equal importance) to 9 (extreme preference or absolute importance) (Table 27). The information from key actors was used to derive the relative importance of one criterion to another using the AHP (Saaty, 1980; Saaty and Vargas, 2000), which has been used in a pairwise comparison technique to assign individual parameter weights for each factor (Table 28).

The individual suitability maps of the selected criteria were classified into four levels of suitability classes and weighed by their importance. The total score for each alternative was calculated by multiplying the importance of weight assigned to each attribute by the scaled value given for that attribute to the alternative and then summing the product's overall attributes (Cengiz and Akbulak 2009; Zardari *et al.*, 2015; Pramanik 2016: Pande et al., 2021a). Thus, all the layers were overlaid by recognizing cell values to the same scale, giving a weight value to the individual criterion, and integrating the weight cell values as given below:

$$\boldsymbol{LS = \sum_{i=1}^{n} Wi\ Xi}\text{………………………….}\mathbf{(17)}$$

Where LS indicates the total land suitability score, Wi denotes the weight assigned to each selected land suitability criteria; Xi indicates the factor score (cells) of i land suitability criteria, and n denotes the total number of land capability criteria.

Table 27*:* Pairwise comparison of the nine-point rating scale after Saaty and Vargas (2000)

Importance	**Definition**	**Explanation**
1	Equal significance	Two criteria contribute equally to the objective criteria
3	Moderate significance	Judgments and experience slightly favor one criterion over another
5	Strong significance	Strongly favored by judgments and experience
7	Very strong significance	A criterion is strongly favored, and its dominance established in practice
9	Extreme importance	The evidence favoring one criterion over another is of the highest probable order of

		affirmation
2,4,6,8,	Intermediate values	When adjustment is needed
Reciprocals	Values for inverse comparison	

The consistency of the judgment matrix is examined with the calculation of the consistency index (CI) as defined by the equation (Mansouri, 2014):

$$\mathrm{CI} = \frac{(\lambda\mathrm{max} - \mathrm{n})}{(n-1)} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\mathbf{(18)}$$

Where CI is the consistency index, λ_{max} is the most significant or principal eigenvalue of the matrix that could be calculated from the matrix and n is the order of the matrix (Ying et al., 2007).

Also, in the AHP, a measure of consistency ratio (CR) was calculated to indicate the randomized probability of matrix judgments as follows:

$$CR = \frac{CI}{RI} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\mathbf{(19)}$$

Where CI is the average of the resulting consistency index, depending on the order of the matrix given by Malczewski (1999; 2000), and RI (Random Index) values for matrices of different sizes (Saaty, 2003).

Table 28: Land suitability criteria classification for apple cultivation

Category	Criteria	Unit	Land suitability classes				Eigenvector weight	Significance (%)	References
			S1 Highly suitable	S2 Moderately suitable	S3 Marginally suitable	S4 Not suitable			
Topography	Elevation	MASL	2500-3000	3000-3400 and 2100-2500	1800-2100	<1800 and >3400	1.51	12.73	Devkota (1999); Manandhar et al. (2014); Getachew *et al.*, (2022)
	Slope	%	<10	10-18	18-28	>28	0.38	5.09	Finnigan et al. (2000); Roots of Peace (2008); Manandhar et al. (2014); Madrigal-Martínez and Puga-Calderón (2018); Getachew *et al.*, (2022)
	Aspect	Direction	East	South, southeast	West, Southwest	North, northwest, northeast	0.19	3.64	Roots of Peace (2008); Manandhar et al., (2014).
Climate	Rainfall	Mm	1000–1500	1500-2000	500-1000	<500 and >2000	0.94	12.73	Chada (2009); Getachew *et al.*, (2022)
	Mean	°C	18-19	17-18 and	16-17 and	<16 and >21	1.41	12.73	Chada (2009);

	Temperature			19-20	20-21				Chattopadhyay (2009); Sharma et al. (2013);
Soil	Soil pH	-log (H^+)	6.0–6.5	5.7–6.0	5.6–5.7	<5.6 and >6.5	0.56	8.49	Getachew *et al.*, (2022)
	Soil Drainage	%	Well-drained and good aeration	Moderately drained and medium aeration	Prone to waterlogging	Waterlogged	0.52	6.37	Chada (2009); Manandhar et al. (2014); Karma and Gibji (2021).
	Soil type	Texture	Sandy loam and sandy clay loam	Silt loam	Loamy sand and clay loam	Gravelly loamy sand, sand, silty clay loam, and clay	1.98	25.47	Roots of Peace (2008); Manandhar et al. (2014); Madrigal-Martínez and Puga-Calderón (2018).
Land management	Land Use/Land Cover	Type	Orchards, tars, and level terraces	Grassland Agricultural land	Forest	Settlement, waterbodies	1.51	12.73	Manandhar et al., (2014); Getachew *et al.*, (2022); Karma and Gibji (2021).

6.4 Results

6.4.1 Suitability of evaluation criteria

The ranking of the nine evaluation criteria showed the importance of each criterion in the order of soil type, elevation, LULC, temperature, rainfall, pH, soil drainage, slope, and aspect (Table 29). Soil type received the highest weight (1.98) followed by elevation and LULC (1.51). The mean temperature, rainfall, soil pH, and soil drainage weight were 1.41, 0.94, 0.56, and 0.52 respectively. Whereas the slope and aspect weighted the lowest at 0.38 and 0.19 respectively. The major soil types in both watersheds are loam and clay. Thus, about 64% and 36% are highly and moderately suitable for the apple tree in the Dire watershed, while about 73.5% and 26.5% are highly and moderately suitable in the Legadadi watershed. The highly suitable elevation of 2500-3000 m.a.s.l constitutes 41.9% and 29.3%; moderately suitable elevation constitutes about 58.1% and 70.7% in the Dire and Legadadi watersheds. On the other hand, 57% and 89% of the watersheds are not suitable for apple growth with the current land use practices in the areas. The mean annual temperature in the Dire and Legedadi watersheds was 15.19°C and 15.02°C in 2020. Thus, favorable mean annual temperature constitutes about 19.59% and 22.79% of the Dire and Legedadi watersheds respectively.

The annual rainfall in the Dire and Legedadi watersheds was about 1406.1 mm and 1886.4 mm in 2021 respectively. The Highly suitable rainfall constitutes about 22.68% of the Dire and 29.77% of the Legedadi respectively, while moderately and marginally suitable rainfall constitutes about 36.08% and 38.14% in the Dire, and about 34.42% and 35.81% in the Legedadi. The pH value of the study watersheds ranges from 4.4 to 7.9. Thus, the results show that about 84% and 62.15% of the Dire and Legedadi watersheds are unsuitable for apple growth from the soil pH perspective respectively. About 43.3% and 56.7%; of the Dire and 44.65% and

55.35% of the Legedadi watersheds were found highly and moderately suitable respectively which is highly required for the successful growth of the apple tree. With regards to the slope factor, about 38.85% and 34.88% of the Dire and Legedadi watersheds fall under the unsuitable slope category. The aspect factor results showed that the east-facing slope has covered about 26.80% and 20.93% of the Dire and Legedadi watersheds which is highly suitable for apple production, and about 34% of the watersheds have a southeast-facing slope, which is moderately suitable for apple production (Table 30).

Table 29: Evaluation criteria, weights, and area under different land suitability classes

Category	Criteria	Dire (%)				Legedadi (%)			
		S1 Highly suitable	S2 Moderately suitable	S3 Marginally suitable	S4 Not suitable	S1 Highly suitable	S2 Moderately suitable	S3 Marginally suitable	S4 Not suitable
Topography	Elevation	41.9	58.1	0.00	0.00	29.30	70.70	0.00	0.00
	Slope	22.52	21.84	16.80	38.85	15.35	9.30	40.47	34.88
	Aspect	26.80	34.02	26.80	12.37	20.93	33.95	20.47	24.65
Climate	Rainfall	22.68	36.08	38.14	3.09	29.77	34.42	35.81	0.00
	Mean temperature	19.59	32.99	24.74	22.68	22.79	31.63	28.37	17.21
Soil	Soil types	63.92	36.08	0.00	0.00	73.49	26.51	0.00	0.00
	Soil pH	12.62	3.18	0.20	83.99	3.72	16.93	17.21	62.14
	Soil drainage	43.30	56.70	0.00	0.00	44.65	55.35	0.00	0.00

Land management	LULC	37.83	5.02	0.00	57.15	11.21	0.00	0.00	88.79

6.4.2 Land suitability analysis

A total of 305 km^2 of land was computed for apple-based agroforestry practices in both watersheds using the criteria established. Out of this about 14 km^2 (6.7%) and 12.34 km^2 (13.1%) of land is highly suitable for apple farming in the Legedadi and Dire watersheds respectively. About 29.29 km^2 (13.9%) 19.05 km^2 (20.3%); 53.86 (25.6%) and 20.12 km^2 (21.4%) are moderately and marginally suitable for apple-based agroforestry farming in the Legedadi and Dire watersheds, respectively. However, about 113.35 km^2 (53.8%) and 42.54 km^2 (45.2%) of land are not suitable in the Legedadi and Dire watersheds respectively (Table 30; Figure 17). The proportion of highly suitable land in the Dire watershed is more considerable (13.1%) than in the Legedadi watershed (6.7%). This could be due to the smaller settlement and infrastructure development in the Dire watershed. However, the highly suitable areas were mainly occurring in the higher elevations of both watersheds.

The Dire watershed has more moderate suitable land (20.3%) than the Legedadi watershed (13.9%). These areas are mainly located adjacent to the highly suitable areas in both watersheds. More specifically areas adjacent to the reservoirs, and farmland stretched out to the periphery of the watersheds. The marginally suitable and unsuitable areas are widely observed in both watersheds. They consisted of about 66.6% and 79.4% in the Dire and Legedadi watersheds respectively. Unsuitable areas were in areas where land was used for other development such as the reservoir, intensive farmland, unpaved and paved roads, and congested settlement areas.

Since these areas are already occupied by other land users, we consider these areas inappropriate for apple farming in both watersheds.

Table 30*:* Quantified land suitable for apple-based agroforestry in the Legedadi and Dire Watersheds

Classes	Suitability	Legedadi		Dire	
		Area (km^2)	%	Area (km^2)	%
S1	Highly suitable	14	6.7	12.34	13.1
S2	Moderately suitable	29.29	13.9	19.05	20.3
S3	Marginally suitable	53.86	25.6	20.12	21.4
N	Not suitable	113.35	53.8	42.54	45.2
Total		210	100	94	100

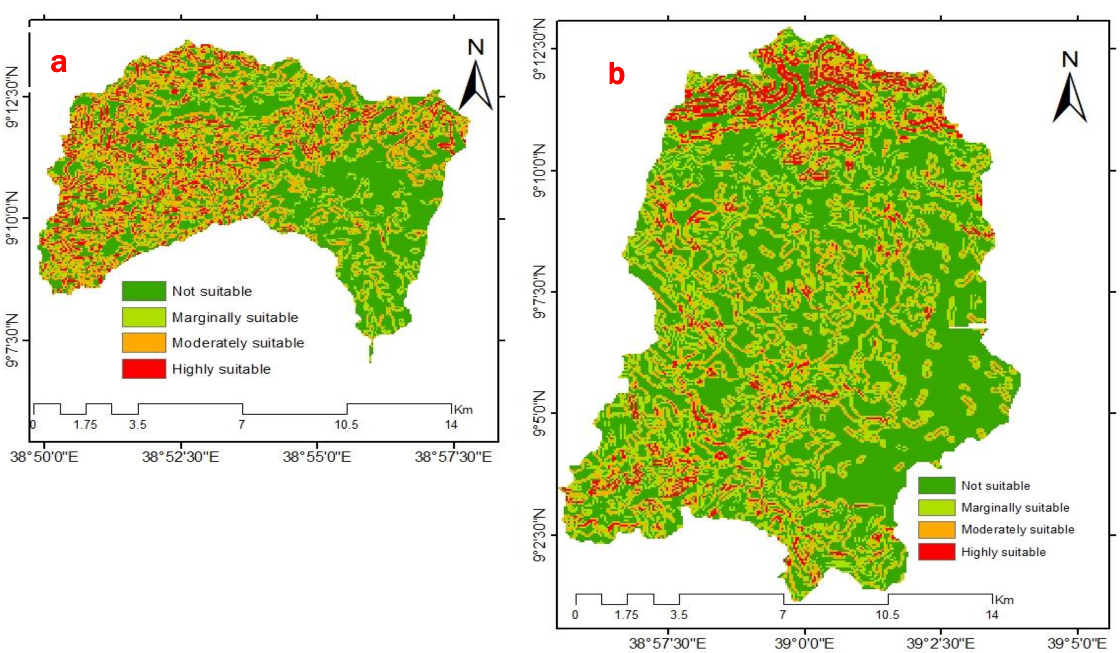

Figure 17: Apple-based agroforestry suitability map in a) Dire watershed and b) Legedadi watershed.

6.5 Discussion

6.5.1 Factors evaluation

Soil is one of the important parameters that determine plant growth (Mustafa *et al.*, 2011). Apple growth can be influenced by soil types. Clay loam and clay soil characteristics are classified as the most suitable and moderately suitable soil textures for apple production (Roots of Peace, 2008; Manandhar et al., 2014; Madrigal-Martínez and Puga-Calderón, 2018). The loam and clay soil types in the watersheds provide higher suitability for apple growth watersheds. Altitude is also an important factor affecting plant growth (Bozdag et al., 2016). High altitudes characterized by lower average temperatures allow faster chilling conditions (Osborne, 2000). Apple trees can be cultivated at an elevation ranging between 1800 and 3000 m.a.s.l (Karma and Gibji, 2021; Getachew et al., 2020); but the altitude ranges between 2500 to 2700 m.a.s.l is highly suitable (Getachew et al., 2020) as the crop gets its chilling requirement to overcome its dormancy (Fadon et al., 2020). The northern parts of the watersheds are the highest peaks which are suitable locations to fulfill the chilling requirement of the tree.

The natural vegetation has been replaced by settlement and agricultural land uses in both watersheds; particularly in the Legedadi watershed (Awraris, 2017). Thus, 90% of the Legedadi watershed is unsuitable for apple growth based on the current land use practices. Temperature affects the net carbon exchange, carbon balance, and carbon partitioning in the apple tree (Zavala, 2004; Calderon-Zavala et al., 2004). The apple tree requires a temperate climatic regime for growth and fruit development (Krishna, 2014). Low temperature is the most significant factor in affecting dormancy completion (Ramirez and Kallarackal, 2014). Temperature also influences the photosynthesis and respiration processes of the apple (Hester and Cacho, 2003). The result revealed that the moderate temperature of the study watersheds could be the most favorable for the apple tree.

The other most important factor affecting the growth of apple trees is rainfall. Both excessive and lower rainfall are harmful to apple growth. Although, the annual rainfall in the Dire (1406.1 mm) and Legedadi (1886.4 mm) watersheds is a little bit over the optimum range for apple production of 1000–1250 mm (Randev, 2009); the entire watersheds are suitable for apple farming. A wide range of soil pH could be favorable for apple trees; however, the highly suitable soil pH ranges between 6.0 and 6.5 (Anderson, 2015). The soil pH in the study area ranges between 4.4 to 7.9 which resulted in most of the watersheds being unsuitable for apple trees. Well-drained soils are considered the best for apple cultivation (Chada, 2009; Chattopadhyay, 2009). Thus, the soil drainage in the study watersheds is well-drained which is favorable for the cultivation of apples.

The slope is also an important indicator of land suitability since it influences drainage, irrigation, and soil erosion (Wu et al., 2009). An increase in slope degree slows down the development of soils and decreases soil depth and fertility (Atalay, 2006). Aspect factors determine sunlight availability for the growth of a particular crop. Thus, aspect is an important factor that affects apple yield (Aggelopoulou et al.,2010). Further, the quality of fruit is greatly determined by the availability of sunlight (Lakiso, 1980; Morales-Quintana et al., 2020). Thus, the availability of suitable aspect conditions could be important for apple growth in the study watersheds.

6.5.2 Land suitability

Sustainable watershed management interventions are vital to improving ecosystem services through restoring ecological integrity and maintaining the biodiversity resources of important landscapes. The practice of agroforestry could also help to improve the well-being of societies at all levels (Jose, 2009). Agroforestry has been recognized as an invaluable tool to control soil erosion by reducing the rate of soil runoff as canopy interception increases (Tyagi *et al.,* 2002) and can retain a large volume of rainfall (Bundela, 2007). The higher infiltration capacity of trees considerably improves the recharge of groundwater, the perennial flow of the stream, and enhanced soil moisture availability through micro-climatic interventions.

The land policy of Ethiopia prohibits practicing farming lands steeper than 30% slope (Lakew, 2005). Thus, the growing of perennial crops (e.g., apple trees) could be invaluable not only for improving ecosystem services but also for ensuring food security. Ranjith *et al.,* (2002) reported that agroforestry combined with contour strips has a significant effect on maintaining soil organic nutrients. The inclusion of trees in soil conservation and erosion control is one of the most widely acclaimed and compelling reasons for including trees on farmlands prone to erosion hazards. Sustainable watershed management practices have generated promising improvements in farm incomes and food security in some of the piloted watersheds in the Tigray, Amhara, and Oromia regions of Ethiopia (Gebreselassie *et al.,* 2016). Therefore, effective, and participatory watershed management undertakings are essential to alleviate the complex socio-ecological problems in the catchment.

Considerable land is suitable for apple farming practices in the Legedadi and Dire watersheds. Therefore, the application of apple farming as a watershed management intervention could significantly contribute to improving water quality and quantity by largely reducing silt in the water reservoirs. As the city of Addis Ababa is highly reliant on ecosystem services (e.g., water provision) from the surrounding central highlands of Ethiopia, water availability is becoming the most important intensively growing constraint in the city (Anteneh *et al.,* 2019). The unsustainable land use practice in the upper catchment has increased soil erosion and significant sediment load to the water reservoirs; for example, in the Legedadi alone the annual soil loss was estimated between 69.9 and 138.4 tons/ha (Gebreegziabher *et al.,* 2021). The annual siltation rate of the reservoir was estimated at 110,500 m^3 and reduced the capacity by 2.1 MCM between 1979 and 1998 (AAWSA, 2011; AAWSA, 2016).

Moreover, Zerihun *et al.,* (2019) reported that the apple-based agroforestry system largely improved the livelihoods of small-scale farmers in the drought-prone areas of north-western Ethiopia. Girmay *et al.,* (2014) indicated that apple grower farmers in southern Ethiopia are earning an average of US$ 272.89 yr^{-1}. Similarly, the practice of apple-based agroforestry farming has contributed to higher net revenue for apple growers in the Dendi district of West Shoa Ethiopia (Lemlem *et al.,* 2013). More recently, Tesfaye and Amene (2022) reported that the livelihood status of apple-producer farmers in the Chencha woreda of southern Ethiopia has greatly improved compared to the previous living conditions. Therefore, with the presence of a huge unmet market demand for apple fruits, apple growers can make an additional income to improve their livelihood. Thus, properly designed apple farming could play a critical role in the improvement of the overall socio-economic and ecological conditions of farmers and reduce their vulnerability to climate-related natural disasters.

6.6 Conclusion

The study area is the most important headwaters of Addis Ababa city which suits ecosystem services that are vital for the inhabitants of Addis Ababa and surrounding communities are benefiting. The growing apple tree has been practiced by AAWSA along the buffer areas of water reservoirs without assessing the land suitability for the crop, thus this study will provide information on apple-based agroforestry farming in the study watershed to support landscape restoration endeavors by key actors in the study watersheds. The result revealed that considerable land is suitable for apple-based agroforestry farming. Thus, the apple tree can be used as a critical watershed management tool to halt the rapid deterioration of the watershed ecosystem. A sustainable watershed management scheme will provide multitudes of benefits; such, first, it contributes to watershed services through improving agroecosystems; second, enhances the local economy through increased income; and third contributes to household food security and wellbeing. However, landholders who play a pivotal role in sustainable watershed management practices should be incentivized to grow economic highland fruits (e.g., apple tree) further sustainable and efficient value-adding supply chain needs to be created to enhance environmental income to households and thereby contribute to alleviating poverty. Although the study demonstrates that apples can be used as a tool to improve watershed services; further investigations are required to improve the harvest of quality apple fruit such as identifying key site-specific factors including soil amenability, electrical conductivity (EC) of soil and water, wind speed during pollination, and the amount of lime in the soil, selecting appropriate apple varieties that are compatible for cross-pollination, trees healthiness, fruit thinning and local pests and diseases are amongst the most important factors that determine the success of apple farming.

Chapter Seven

7 Synthesis

The big upper Akaki catchment is not only the prime source of water but also an important contributor Awash River basin (AAWSA, 2016; Simeneh et al., 2023a); their hydrological function is essential for socioeconomic development in the Basin (Simeneh et al., 2023). Unlikely, the rapid conversion of natural areas into settlements and farmland (AAWSA, 2016; Awraris, 2017; Simeneh et al., 2023a) has reduced the ecosystem service value of the landscapes (Simeneh et al., 2023a) and deteriorated ecosystem quality (Simeneh et al., 2023b). These land use modifications affect the hydrology of the watersheds (Sisay, 2014; Ajanaw, 2021; Simeneh et al., 2023c), and the welfare of downstream water users (Yilikal et al., 2019). Furthermore, greatly increased soil erosion and sediment load to the reservoirs (Gebreegziabher et al., 2021). Meeting the daily water demand of the city requires a supply of 0.8 MCM to 1.2 MCM (Van Rooijen and Taddesse, 2009; Tirusew and Fekadu, 2023).

The chronically unbalanced water supply and demand cannot satisfy the current fast-growing population (Tirusew and Fekadu, 2023). Further, Addis Ababa's competitiveness as a diplomatic hub, business, and industrial center could be negatively impacted. Unregulated groundwater extraction can be a short-term solution but not sustainable and affects groundwater mismanagement (Tirusew and Fekadu, 2023). The city dwellers are not satisfied with the water services (Yilikal et al., 2019), as water demand has remained suppressed, with a current supply-demand gap (World Bank, 2007; Tekle, 2008). To improve water supply, inhabitants of major cities of Ethiopia including Addis Ababa residents have demonstrated their support for

Ecosystem-Based Water Supply Management (EBWSM) through direct payment (Yilikal et al., 2019; Solomon et al., 2019).

Watershed management practice is dynamic (Getahun, 2022) that involves the management of land, water, and biota, for socioecological and economic developments (Wang et al., 2016). Community-based integrated watershed management program was introduced about two decades ago to tackle natural resource depletion and improve the livelihood of Ethiopians (Gebregziabher *et al.*, 2016; Battistelli et al., 2022); however, most of them were not successful (Tesfaye et al., 2014). The rapid and quick-time cognizance of farmers has hindered the program (Gebreselassie et al., 2009). Watershed management involves serval features (Gebregziabher et al., 2016) and requires a long time to get the benefits (Hagos et al., 2018; Battistelli et al., 2022; Kissinger *et al.*, 2013).

Farmers in the area have a limited understanding of the advantages of sustainable watershed management practices (Hana, 2020). Although Ethiopian land policy prohibits farming on slopes steeper than 30% (Lakew, 2005), they are engaged in conventional farming practices. Growing annual plants on steep slopes is popular due to a scarcity of farmland and a lack of viable livelihood options in the area. Furthermore, inadequate financing is the foremost constraint to reverse land degradation in Ethiopia (Zeleke, 2017; Battistelli et al., 2022). Therefore incentive-based conservation approaches are required (Battistelli et al., 2022) that can offer diversified as well as increased funding for accelerating conservation in Ethiopia (Battistelli et al., 2022). For example, in the Bale Mountains Ecoregion REDD+ Project, the first-ever carbon credit sale was hosted despite the lack of a legal framework (Farm Africa, 2022). Similarly, several PES projects, including the Choke, Arjo-Diga woodland, Hadew rangeland, and Kulfo watershed, are currently being piloted (Admanaban and Fekadu, 2017).

Therefore, this research can be used as the primary basis of information to help policymakers and researchers understand the need for sustainable watershed management intervention to support nature conservation through long-term financial investments in the central watersheds and elsewhere. Several economic instruments have been broadly implemented in numerous growing nations to reverse the ecosystem degradation traits by way of linking upstream with downstream by addressing the environmental externalities that affect downstream users (Fisher et al., 2010; Nepal et al., 2014). Among the various instruments, the PES is the most applicable and sustainable financing mechanism (Greiber, 2009; FAO, 2014; Rode et al., 2016). PES is rooted within the demand for ecosystem services and entails people outside the range (Farley and Costanza, 2010). Since PES affects land use decisions by enabling landholders to enhance environmental services; landholders need to recognize the tradeoff between current and alternative farming practices (Wegner, 2015; Nyongesa, 2011; Rode et al., 2016). The ability of land users to transition to sustainable land practices is heavily influenced by a socio-economic situation (Wegner, 2015). Payments are frequently made using the opportunity cost rather than the monetary value (Jack et al.,2008). Additional revenue streams include regenerative agriculture practices (Fripp; 2014; Wollenberg et al., 2022), receiving payments for controlling nutrient runoff (Okiria et al., 2021), and eliminating pesticide use (Niklas and Robert, 2022) can be obtained. User-financed PES programs were more closely adapted to local conditions and needs in both developed and developing countries than government-financed PES programs (Wunder et al., 2008).

PES is an important tool for managing shared resources (Fisher et al., 2010). It functions as fixed social rules, which may be the result of the collective governance model (Murtinho and Hayes, 2017). PES has long been used as a critical ecosystem conservation and livelihood enhancement tool in neighboring Kenya (Nyongesa, 2011). Ethiopia recently emphasized the importance of incentive-based conservation practices in its National Adaptation Plan (FDRE, 2019). Although it is late, the use of economic instruments is invaluable and necessary to replace outdated and conventional command and control methods (Hagos et al., 2011). To address chronic water poverty, a multitude of solutions would be required; however, improving the management of existing watersheds through market-based inventive instruments is also vital (Yilikal et al., 2019). Although successful water-based payment programs were implemented in the global north; successful WPES exist in important cities of the global south such as Cuenca in Ecuador and Rio de Janeiro in Brazil (TNC, 2015), and the Upper-Tana- Nairobi Water Fund (UTNWF) in Kenya is the first of its kind in Africa (TNC, 2015).

Mapping the important elements of sustainable watershed management as well as designing new ways to fund conservation interventions is critical (Battistelli et al., 2022). Furthermore, a detailed assessment of sellable ecosystem services; market demand; and the economic impact of land use change needs to be defined before project implementation (Langat *et al.,* 2017). The development of robust ecosystem service monitoring systems and standardized impact assessment tools are also important to detect changes in the ecosystem after watershed management interventions are implemented (Wunder, 2006; Brouwer et al., 2011) and information needs to be accessed by key parties to ensure accountability and long-term PES program. Those who pay must engage in meaningful activities to ensure the long-term supply of ecosystem services (Greiber, 2009). Land security was critical to their successful payments for watershed services (PWS) schemes in developing countries (Porras et al., 2008). Most importantly, the effective participation of local governments in the implementation of PES systems is critical for their acceptance by local communities (Brouwer et al., 2011; Young and de Bakker, 2014).

Legal and regulatory frameworks that provide clarity on governance are required for successful PES schemes (Greiber, 2009; FAO, 2016; Gebreselassie et al., 2016). In contrast, the absence of supportive legal frameworks is a major impediment to the successful implementation of PES in many countries (Greiber, 2009). Legal interventions in PES also provide facts and clarity for PES actors and inspire participation. This is beneficial as public entities can invest in public funds or use public goods, and PES-related legislation creates legal certainty and develops trust among parties (Greiber, 2009). The sociocultural context, including an appropriate distribution of natural resource rights and obligations, as well as ethical considerations, should be critically examined (Rode et al., 2016). In contrast to unilateral decision-making processes (Maren et al., 2014; Memon and Thapa, 2016), a collective decision on the PES can improve the provision and regulation of ecosystem services on the supply side (Adhikari and Agrawal, 2013; Sattler et al., 2015).

Despite the government's promising interest in environmental protection, legal frameworks that allow for the incorporation and design of compatible incentive-based conservation mechanisms remain lacking. Prior land certification experience has increased tenure security and conservation finance (Holden et al., 2009; Hagos, 2012). The proclamation on community-based watershed management and utilization (Federal Negarit Gazette, 2020) can be viewed as an important step toward watershed management interventions. Further, the recently adopted capital market proclamation is expected to deliver various reforms (Federal Negarit Gazette, 2021) this could help to tap into new funding to protect the natural ecosystems (Battistelli et al., 2022).

Identifying and defining the role and interest of key actors within the PES scheme is exceptionally vital. Moreover, a depth analysis of existing governance systems inside the designing of the PES scheme is required. According to (World Bank, 2008), public institutions, CBOs, NGOs, and financial institutions that provide resources are vital actors in the PES scheme. However, the capacity to implement a PES scheme requires an understanding of existing information and interests (Simelton *et al.*, 2013). Likewise, the key stakeholders of the PES programs include the Addis Ababa Water and Sewage Authority, Community-Based Organizations (CBOs), private actors can be private ventures that operate in both watersheds can be PES project implementing agencies as they are the ecosystem sellers and buyers. while regulatory agencies including but not limited to the Oromia national regional state bureaus of Agriculture and Natural Resources, Rural Land Administration and Use, and the Addis Ababa City Administration Environmental Protection Authority are the important actors for the PES project. Nongovernmental intermediaries include organizations such as weAspire, Vitens Evides International, UN-Habitat, the World Bank, and the United Nations Development Programme (UNDP). Addis Ababa University, Kotebe Metropolitan University, and Oromia Agricultural Research Institute are among the research and academic institutions.

The sustainable watershed management scheme is expected to increase environmental profits (Joram *et al.*, 2018) and thus contribute to poverty alleviation. For example, erosion reduction leads to more favorable soil properties in terms of soil fertility and water retention (Springgay, 2019). Improved agricultural land and inputs could help to enhance water supply and quality (Tesfaye et al., 2016). The provision of freshwater and the improvement of water quality and quantity and flow regulation are vital benefits obtained from watersheds that determine the social and economic well-being of downstream users (Springgay, 2019).

Agroforestry practices play an important role in natural resource conservation. In the study area, a significant amount of land was discovered to be suitable for apple-based agroforestry farming (Simeneh et al., 2022). Furthermore, the watersheds have an unparallel opportunity for ecotourism development (Simeneh et al., 2023e); therefore, ecotourism can be also used as an alternative financing mechanism that can boost landscape biodiversity and the local economy (Simeneh, 2020).

Catchment investment programs in Africa have demonstrated greater conservation yields a nine to 23-fold increase in return (Bennett and Carroll, 2014; TNC, 2016). Further, the Upper Tana water fund program has generated a viable return on investment which includes a discount in sediment awareness and a boom in annual water and agricultural yields (TNC, 2015). Ethiopian land policy prohibits farming on slopes steeper than 30% (Lakew, 2005), however, most farmers are engaged in conventional farming practices. Thus, contributing to soil nutrient depletion; the loss from agricultural lands resulted in a crop production loss of 104 million tonnes with a market value of USD 48.35 billion between 2003-2016 (Tilahun, 2020).

The total annual ecosystem service value in the Legdadi was estimated between US$ 11.9 million and US$ 9.66 million while it was estimated between US$ 59 thousand and US$ 36 thousand in the Dire watershed (Simeneh *et al.*, 2023a). Furthermore, using information obtained from Van der Ploeg et al., (2010) and cascaded into the regional coefficient of ESV function (McVittie and Hussain, 2013), the annual water-related ecosystem services within the Legedadi and Dire watersheds were predicted to be US$ 43.6 thousand and US$ 15.4 thousand, respectively (Simeneh et al., 2023a). The construction of sediment traps was proposed to help reduce sedimentation in both watersheds at a total estimated cost of US$13.5 million in the

Legedai watershed and US$1.1 million in the Dire watershed, respectively (AAWSA, 2011), which appears to be an unwise investment.

7.1 Conclusion

Both watersheds are the most important headwaters which provide suits of ecosystem services. Thus, they are also critical to the overall socioeconomic development. However, the anthropogenic disturbance rapidly deteriorates the entire ecosystem, resulting in a decrease not only in ecosystem service values but also in the quality of habitat in both watersheds. The decline in ecosystem service values and quality of habitat is expected to continue. Although water yield has increased in both watersheds due to climate variability, a decrease in water yield may occur in the future. Despite land use changes having a lower impact on water yield in both watersheds, unsustainable land management practices may lead to natural hazards and make downstream people susceptible to water shortage.

The presence of diverse water bird species in the watersheds can make the area better for ecotourism development and biodiversity conservation. A proper ecotourism development can generate revenue for local communities while also raising conservation awareness. Thus, ecotourism can be used as a landscape management tool to take economic advantage of the ever-increasing demand for outdoor recreational activities in watersheds. Furthermore, agro-trees can be used to restore the watershed ecosystem.

Although further detailed study is required to gain an actual understanding of the complex ecosystem services in the watersheds; the findings of this study could be used for planning and management intervention including reducing land degradation and climate change risks, recreational services development, increasing agricultural productivity, improving livelihoods, and a wider range of economic and environmental benefits in the larger watersheds and beyond.

7.2 Recommendations

Designing innovative financing mechanisms (e.g., PES, water fund, etc.) is therefore highly critical to abate biodiversity loss and improve the local economy. In addition to the global framework, the study can be linked to national and sectoral level strategies such as the National Adaption Plan (NAP), Nationally Determined Contributions (NDCs), National Adaption Plan (NAP-ETH), and Climate Resilient Green Economy (CRGE) all of which have evolved to support efforts to address climate change risks. Payment for ecosystem services has enormous potential for use as a tool to address complex socioeconomic and environmental issues in the watersheds and at large. The foremost reasons that make PES feasible could include; first, the rapid deterioration of watersheds, as they are losing their ability to provide water services; second, hydrological instability and water poverty could result in a greater negative socioeconomic impact and well-being in the region; third, downstream water users have demonstrated their willingness to support the Ecosystem-Based Water Supply Management program; forth, the government has been proposing multi-million dollar unsustainable sediment trapping dam projects; and finally, the local community has recognized the importance of sustainable land use.

- Therefore, designing incentive-based approaches to reverse biodiversity decline is critical. However, further research on the prospects of the instruments needed to be assessed.
- An urgent and active watershed rehabilitation program is required with the active participation of key stakeholders.

- Given the increasing enthusiasm for outdoor leisure pursuits in the vicinity, judicious ecotourism strategizing stands to yield dividends for the indigenous populace.
- Furthermore, the presence of such a diverse bird species may make the area valuable enough to be designated as an important bird area, which could support effective biodiversity conservation and hasten the implementation of a globally agreed biodiversity framework.
- Ecotourism planning and management interventions need to be designed to improve the recreational value of the landscapes.
- Smallholder farmers should be incentivized to cultivate high-value fruits suited to highland ecosystems.

References

Abhijit M. Zende., R. Nagarajan., and K. R. Ata (2013). Prioritization of sub-watersheds in a semi-arid region, Western Maharashtra, India using Geographical Information System. American Journal of Engineering Research (AJER); 02:10;128-135.

Abiye, T. A., Sulieman, H., and Ayalew, M. (2009). Use of treated wastewater for managed aquifer recharge in highly populated urban centers: a case study in Addis Ababa, Ethiopia. Environmental Geology, 58: 55-59.

Abreham Berta, Teshome Soromessa and Eyasu Elias (2019). The effect of land use land cover changes on ecosystem services valuation of Winike watershed, Omo Gibe basin, Ethiopia. Human and Ecological Risk Assessment: An International Journal, 26:10, 2608-262.

Abreham Berta, Tomasz Noszczyk, Teshome Soromessa and Eyasu Elias (2020). The InVEST Habitat Quality Model Associated with Land Use/Cover Change in the Winike Watershed in the Omo-Gibe Basin, Southwest Ethiopia. Remote Sens; 12: 1103.

Addis Ababa Water and Sewerage Authority (AAWSA) (2011). Master plan review, catchment rehabilitation, and awareness creation for Geffersa, Legedadi, and Dire catchments. Addis Ababa, Ethiopia.

Addis Ababa Water and Sewerage Authority (AAWSA) (2016). Urban water supply and sanitation; Legedadi and Dire dam rehabilitation project; Addis Ababa, Ethiopia.

Adem Esmail, B., Cortinovis, C., Wang, J., Genletti, D., and Albert, C (2023). Mapping and assessing ecosystem services for sustainable policy and decision-making in Eritrea. *Ambio*.

Adhikaria Bhim and Agrawalb Arun (2013). Understanding the social and ecological outcomes of PES projects: A review and an analysis. Conservation and Society 11(4): 359-374.

Admanaban Murugan and Fekadu Israel (2017). Impact of Forest Carbon Sequestration Initiative on Community Assets: The Case of Assisted Natural Regeneration Project in Humbo, Southwestern Ethiopia. African Studies Quarterly, 17 (1).

Adugna Feyissa Gubena (2016). Environmental Impact Assessment in Ethiopia: A General Review of History, Transformation and Challenges Hindering Full Implementation. Journal of Environment and Earth Science Vol.6, No.1.

Aggelopoulou, K. D., Wulfsohn, D. Fountas, S. Gemtos, T. A. Nanos, G. D. and Blackmore S. (2010). Spatial variation in yield and quality in a small apple orchard," *Precision Agriculture*, vol. 11, no. 5, pp. 538–556.

Aghsaei, H.; Dinan, N.M.; Moridi, A.; Asadolahi, Z.; Delavar, M.; Fohrer, N.; Wagner, P.D (2020). Effects of dynamic land use/land cover change on water resources and sediment yield in the Anzali wetland catchment, Gilan, Iran. Sci. Total Environ. 712, 136449.

Agidew, A. A. (2015). Land suitability evaluation for sorghum and barley crops in South Wollo Zone of Ethiopia. Journal of Economics and Sustainable Development, 6(1), 14–26.

Agroforestry Network (2018). Scaling up Agroforestry: Potential, Challenges and Barriers. A review of environmental, social, and economic aspects on the farmer, community, and landscape level.

Aguilar, R., Quesada, M., Ashworth, L., Herrerias, Diego Y., and Lobo, J (2008). Genetic consequences of habitat fragmentation in plant populations: Susceptible signals in plant traits and methodological approaches. Molecular Ecology17:5177–5188.

Ajanaw Negese (2021). Impacts of Land Use and Land Cover Change on Soil Erosion and Hydrological Responses in Ethiopia, Applied and Environmental Soil Science, vol. 2021, Article ID 6669438, 10 pages.

Alarcon, G.G.; de Freitas, L.A.; dos, S.; da Fountoura, G.O.; de Macedo, C.X.; Riberio, D.C (2016). The challenges of implementing a legal framework for payment for ecosystem services in Santa Catarina. Braz. J. Nat. Conserv. 2, 132–136.

Albert, C., Aronson, J., Fürst, C., and Opdam, P. (2014). Integrating ecosystem services in landscape planning: requirements, approaches, and impacts. *Landscape Ecology, 29(8), 1277–1285.*

Allen, R.G., Pereira, L.S., Raes, D. and Smith, M., (1998). "Crop evapotranspiration. Guidelines for computing crop water requirements." FAO Irrigation and Drainage Paper 56. Food and Agriculture Organization of the United Nations, Rome, Italy.

Allen, R., Pruitt, W., Raes, D., Smith, M. and Pereira, L. (2005). "Estimating Evaporation from Bare Soil and the Crop Coefficient for the Initial Period Using Common Soils Information." Journal of Irrigation and Drainage Engineering, 131(1): 14-23.

Almpanidou V, Mazaris AD, Mertzanis Y, Avraam I, Antoniou I, Pantis JD, Sgardelis SP (2014). Providing insights on habitat connectivity for male brown bears: a combination of habitat suitability and landscape graph-based models. Ecol Model 286:37–44

Amici, V., Rocchini, D., Filibeck, G., Bacaro, G., Santi, E., Geri, F., Landi, S., Scoppola, A., Chiarucci, A., (2015). Landscape structure effects on forest plant diversity at local scale: exploring the role of spatial extent. Ecol. Complex 21, 44–52.

Amsalu Aklilu., Stroosnijder, L., and Graaff, J. De (2007). Long-term alterations in land resource use and the driving forces in the Beressa watershed, highlands of Ethiopia. Journal of Environmental Management, 83(4), 448–459.

Anderson, P. R. (2015). Low-chill apple cultivars for North Floridaand North Central Florida.

Annandale, George W.; Morris, Gregory L.; Karki, Pravin (2016). Extending the Life of Reservoirs: Sustainable Sediment Management for Dams and Run-of-River Hydropower. Directions in Development--Energy and Mining; Washington, DC: World Bank.

Arabinda L (2003). Integrating GIS and multi-criteria decision-making techniques for land resource planning. M.S. Thesis. International Institute for Geo-Information Science and Earth Observation. Enschede: The Netherlands.

Aramde Fetene, Thomas Hilker, Kumelachew Yeshitela, Ruediger Prasse, Warren Cohen and Zhiqiang Yang (2016). Detecting Trends in Land use and Land cover Change of Nech Sar National Park, Ethiopia. Environmental Management, 57:137–147.

Arowolo, A.O.; Deng, X.; Olatunji, O.A.; Obayelu, A.E (2018). Assessing ecosystem service values changes in response to land use and land cover dynamics in Nigeria. Sci. Total Environ. 636, 597–609.

Aschonitis, V.G.; Gaglio, M.; Castaldelli, G.; Fano, E.A. (2016). Criticism on elasticity-sensitivity coefficient for assessing the robustness and sensitivity of ecosystem services values. Ecosystem Services, 20: 66–68.

Ashebir D, Deckers T, Nyssen J, Bihon W, Tsegay A, Tekie H, Poesen J, Haile M, Wondumagegneheu F, Raes D (2010). Growing apple (*Malus domestica*) under tropical mountain climate conditions in Northern Ethiopia. Exp Agric 46:53–65.

Ashbir Woldeyohannes., Cotter, M., Webanesh Desslegn and Girma Kelboro (2020). Assessing Changes in Ecosystem Service Values over 1985–2050 in Response to Land Use and Land Cover Dynamics in Abaya-Chamo Basin, Southern Ethiopia. Land; 9(2): 37.

Asmamaw Adamu (2013). Assessing The Impacts of Land Use and Land Cover Change on Hydrology of Watershed: A Case study on Gilgel–Abbay Watershed, Lake Tana Basin, Ethiopia. MSc dissertation. EU-Erasmus Mundus.

Aster, D.Y. (2004). Use of geospatial technology for environmental protection in Ethiopia. Paper presented at a workshop on the use of space technology for natural resource management, monitoring and disaster management. United Nation/European Space Agency/Sudan Remote Sensing Agency, Khartoum, Sudan.

Atalay I. (2006). The effects of mountainous areas on biodiversity: a case study from the northern Anatolin mountains and the Taurus Mountains, *Grazer Schriften der Geographie und Raumforschung*, vol. 41, pp. 17–26.

Awraris Mamo (2017). Analysis of the Impact of Land Use Land Cover Dynamics on Soil Erosion Using the RUSLE Model: The Case of Water Supply Reservoir Watersheds around Addis Ababa. MSc thesis; Addis Ababa University, Addis Ababa.

Bao, S.; Xiao, N.; Lai, Z.; Zhang, H.; Kim, C (2015). Optimizing watchtower locations for forest fire monitoring using location models. Fire Saf. J., 71, 100–109.

Bai, Y.; Ochuodho, T.O.; Yang, J (2019). Impact of land use and climate change on water-related ecosystem services in Kentucky, USA. Ecol. Indic., 102, 51–64.

Baral Himlal, Rodney J. Keenan, Sunil K. Sharma, Nigel E. Stork and Sabine Kasel (2014). Spatial assessment and mapping of biodiversity and conservation priorities in a heavily modified and fragmented production landscape in north-central Victoria, Australia. Ecological Indicators, 36:552– 562.

Basane Claire Ntshane and James Gambiza (2016). Habitat assessment for ecosystem services in South Africa, International Journal of Biodiversity Science, Ecosystem Services and Management, 12:4, 242-254.

Battistelli, F., J.A. Tadesse, and L. Marsters. (2022). “Financing Sustainable Watershed Management in Ethiopia: Exploring Innovative Financing Strategies for Nature-Based Solutions.” Working Paper. Washington, DC: World Resources Institute.

Behailu, A., and Kebede, J., (2018). A study on the causes of apple (Malus Domestica) fruit loss at Chencha Woreda of Gamo Gofa Zone, Southern Ethiopia. J. Stored Prod. Postharvest Res. 9 (7), 72–77.

Belay Tatek and Mengistu Daniel (2019). LULC dynamics and drivers in the Muga watershed, Upper Blue Nile basin, Ethiopia. Remote Sensing Applications. Society and Environment, 15:100249.

Belay Tegene. (2002). Land cover/ use changes in the Derekolli catchment of the South Welo Zone of Amhara Region, Ethiopia. Eastern Africa Social Science Research Review 18(1): 1-20.

Benayas, J. M. R., Newton, A. C., Diaz, A. and Bullock, J. M. (2009). Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. Science 325, 1121–1124.

Bennett, A., Saunders, D., (2010). Habitat fragmentation and landscape change. Conserv. Biol. All 93, 1544–1550.

Bennett, G., and N. Carroll. (2014). Gaining Depth: State of Watershed Investment 2014.Available online at www.ecosystemmarketplace.com/reports/sowi2014

Bertram, C. and Larondelle, N (2016). Going to the woods is going home: Recreational benefits of a larger urban forest site-A travel cost analysis for Berlin, Germany. Ecol. Econ. 132, 255–263.

Bezabih A (2009). Growing apple (*Malus domestica*) under tropical mountain climate conditions in Northern Ethiopia. Expl Agric 46: 53-65.

Biedemariam Muluberhan, Emiru Birhane, Biadgilgn Demissie, Tewodros Tadesse, Girmay Gebresamuel, and Solomon Habtu (2022). "Ecosystem Service Values as Related to Land Use and Land Cover Changes in Ethiopia: A Review" *Land* 11, no. 12: 2212.

Bonan, G (2015). Ecological Climatology: Concepts and Applications; Cambridge University Press: Cambridge, UK.

Bower, K.M (2014). Water supply and sanitation of Costa Rica. Environ. Earth Sci. 71, 107–123.

Boyd, J and Banzhaf, S. (2007). What are ecosystem services? Ecological Economics 63 (2–3), 616–626.

Bozdag, A. Yavus, F. and A. S. Gnay, (2016). “AHP and GIS-based land suitability analysis for Cihanbeyli County,” *Environmental Earth Sciences*, vol. 75, p. 813.

Bratman, G. N., Anderson, C. B., Berman, M. G., Cochran, B., de Vries, S., Flanders, J., … Daily, G. C. (2019). Nature and mental health: An ecosystem service perspective. *Science Advances, 5(7), eaax0903.*

Brooks T, Mittermeier R, Fonseca G, Gerlach J, Hoffmann M, Lamoreux J, Mittermeier C, Pilgrim J, and Rodrigues A (2006). Global Biodiversity Conservation Priorities. *J. science* 313. DOI-10.1126/science.1127609.

Brouwer, R., Tesfaye, A., and Pauw, P. (2011). Meta-analysis of institutional-economic factors explaining the environmental performance of payments for watershed services. *Environmental Conservation, 38(04), 380–392.*

Browder G., Ozment S., Rehberger Bescos I., Gartner T., and Lange G-M. (2019). Integrating Green and Gray: Creating Next Generation. Infrastructure. Washington, DC: World Bank and World Resources Institute.

Bruijnzeel, L. A (2004). Hydrological functions of tropical forests: Not seeing the soil for the trees? Agriculture Ecosystems and Environment 104: 185–228.

Bundela, D. S. (2007). Water Management in Northeast India - Some Case Studies. Edited by Kaledhonkar et al. CSSRI, Karnal. pp. 40-46.

Burkhard, B., Kroll, F., Nedkov, S., and Müller, F. (2012). Mapping ecosystem service supply, demand and budgets. *Ecological Indicators, 21, 17–29.*

Bustamante, M.; Robledo-Abad, C.; Harper, R.; Mbow, C.; Ravindranat, N.H.; Sperling, F.; Haberl, H.; de Pinto, A.S.; Smith, P. (2014). Co-Benefits, Trade-Offs, Barriers and Policies for Greenhouse Gas Mitigation in the Agriculture, Forestry and Other Land Use (AFOLU) Sector. Glob. Chang. Biol, 20:3270–329.

Calderon-Zavala, G. Lakso A. N., and Piccioni R. M. (2004). Temperature effects on fruit and shoot growth in the apple (Malus Domestica) early in the season," *Acta Horticulturae*, vol. 636.

Cardinale, B.J., J. Emmett Duffy, Andrew Gonzalez, David U. Hooper, Charles Perrings, Patrick Venail, Anita Narwani, Georgina M. Mace, David Tilman, David A. Wardle, Ann P. Kinzig, Gretchen C. Daily, Michel Loreau, James B. Grace, Anne Larigauderie, Diane S. Srivastava and Shahid Naeem (2012). Biodiversity loss and its impact on humanity. Nature, 486:11148.

Caro Claudia, João Carlos Marques, Pedro P. Cunha, Zara Teixeira (2020). Ecosystem services as a resilience descriptor in habitat risk assessment using the InVEST model. Ecological Indicators.115:106426.

Casado-Arzuaga, I.; Onaindia, M.; Madariaga, I.; Verburg, P.H (2014). Mapping recreation and aesthetic value of ecosystems in the Bilbao Metropolitan Greenbelt (northern Spain) to support landscape planning. Landsc. Ecol., 29, 1393–1405.

Caspersen, O.H.; Olafsson, A.S (2010). Recreational mapping and planning for enlargement of the green structure in greater Copenhagen. Urban For. Urban Green. 9;101–112.

Castillo-Eguskitza, N., Martín-López, B., and Onaindia, M. (2018). A comprehensive assessment of ecosystem services: Integrating supply, demand and interest in the Urdaibai Biosphere Reserve. *Ecological Indicators, 93, 1176–1189.*

Ceballos, G and Ehrlich, P.R (2006). Global mammal distributions, biodiversity hotspots, and conservation. Proc Natl Acad Sci USA 103: 19374–19379.

Cengiz T, and Akbulak C (2009). Application of Analytical Hierarchy Process and Geographic Information Systems in land-use suitability evaluation: a case study of Dümrek village (Çanakkale, Turkey). Int J SustDev World Ecol 16(4):286–294.

Chada KL (2009). Handbook of horticulture, Indian Council of Agriculture Research (ICAR), New Delhi, India.

Chang Zhao (2018). Quantifying and mapping the supply of and demand for urban ecosystem services." PhD thesis, University of Lowa.

Chase T. N., Pielke R. A., T. G. F. KittelR, R. Nemani S, W. Runnin (1999). Simulated Impacts of Historical Land Cover Changes on Global Climate in Northern Winter. Climate Dynamics 16:93-105.

Chattopadhyay T. K. (2009). A Textbook on Pomology (Temperate Fruits), Kalyani Publishers, Chennai, India.

Chen, X., Vierling, L., and Deering, D. (2005). A simple and effective radiometric correction method to improve landscape change detection across sensors and across time. Remote Sensing of Environment, 98(1), 63–79.

Chen, Y., Qiao, F., Jiang, L., (2016). Effects of land use pattern change on regional scale habitat quality based on InVEST modelda case study in Beijing. Acta Sci. Nat. Univ. Pekin. 52, 553–562.

Cheng Zhang, Jing Li, and Zixiang Zhou (2022). "Ecosystem service cascade: Concept, review, application and prospect" *Ecological indicators* 137: 108766.

Chiara Cortinovis, Grazia Zulian and Davide Geneletti (2018). Assessing Nature-Based Recreation to Support Urban Green Infrastructure Planning in Trento (Italy) Land; 7, 112.

Chuma G´eant Basimine, Nad`ege Cizungu Cirezi, Jean Mubalama Mondo, Yannick Mugumaarhahama, Deckas Mushamalirwa Ganza, Karume Katcho, Gustave Nachigera Mushagalusa, Serge Schmitz (2021). Suitability for agroforestry implementation around Itombwe Natural Reserve (RNI), eastern DR Congo:

Application of the Analytical Hierarchy Process (AHP) approach in geographic information system tool. Trees, Forests and People, 6:100125.

Church, A., Fish, R., Haines-Young, R., Mourato, S., Tratalos, J., … Willis, C. (2014). UK National Ecosystem Assessment Follow-on. Work Package Report 5: Cultural ecosystem services and indicators. UNEP-WCMC, LWEC, UK.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K.,Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, G.R., Sutton, P., and van der Belt, (1997). The value of the world's ecosystem services and natural capital. Nature 387; 253-260.

Costanza Robert, Brendan Fisher, Kenneth Mulder, Shuang Liu, Treg Christopher (2007). Biodiversity and ecosystem services: A multi-scale empirical study of the relationship between species richness and net primary production. Ecological Economics 61: 478 – 491.

Crossman, N.D., Burkhard, B., Nedkov, S., Willemen, L., Petz, K., Palomo, I., Drakou, E.G., Martín-Lopez, B., McPhearson, T., Boyanova, K., Alkemade, R., Egoh, B., Dunbar, M.B., Maes, J. (2013). A blueprint for mapping and modelling ecosystem services. Ecosystem Services, 4, 4–14.

Dagnachew Legesse, Christine Vallet-Coulomb, and Francoise Gasse (2003). Hydrological response of a catchment to climate and land use changes in Tropical Africa: case study South Central Ethiopia. Journal of Hydrology; 275: 67–85.

Daily GC, and Matson PA (2008). Ecosystem services: from theory to implementation. Proc Natl Acad Sci U S A. 105(28):9455–6.

de Groot, R.S., Wilson, M.A., Boumans, R.M.J. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological Economics 41, 393-408.

de Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L. (2010). Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological Complexity 7: 260-272.

de Groot, R.; Jax, K. and P. Harrison (2016): Links between Biodiversity and Ecosystem Services. OpenNESS Ecosystem Services Reference Book.

Devkota LN (1999). Deciduous fruit production in Nepal. In: Papademetriou MK, Herath EM (ed) Deciduous fruit production inAsia and the Pacific. RAP Publication: 1999/10. Food and Agriculture Organization of the United Nations: Regional Office for Asia and the Pacific, Bangkok, pp 63-78.

Do, V. H., La, N., Mulia, R., Bergkvist, G., Dahlin, A. S., Nguyen, V. T., Pham, H.T and Öborn, I. (2020). Fruit Tree-Based Agroforestry Systems for Smallholder Farmers in Northwest Vietnam—A Quantitative and Qualitative Assessment. Land, 9(11), 451.

Donohue, R. J., M. L. Roderick, and T. R. McVicar (2012). Roots, storms and soil pores: Incorporating key ecohydrological processes into Budyko's hydrological model, Journal of Hydrology, 436-437, 35-50.

Duarte GT, Ribeiro MC, and Paglia AP (2016). Ecosystem Services Modeling as a Tool for Defining Priority Areas for Conservation. PLoS ONE 11(5): e0154573.

Dwivedi, R.S., Sreenivas, K and Ramana, K.V (2005). Land use land cover change analysis in part of Ethiopia using Landsat thematic mapper data. International journal of remote sensing 26(7): 1285-1287.

Easton, Z.M., (2016). Soil and Soil Water Relationships. Virginia Cooperative Extension Programs. Communications and Marketing, College of Agriculture and Life Sciences, Virginia University, USA.

Edgars Jūrmalis and Zane Lībiete (2020). Developing a Framework for Characterizing Recreational Potential of Forest Areas Using Weighted Criteria Analysis**.** Research for Rural Development 2019, At: Jelgava, Latvia, Volume: 1.

Edmonds, N. J., Al-Zaidan, A. S., Al-Sabah, A. A., Le Quesne, W. J. F., Devlin, M. J., Davison, P. I., and Lyons B.P. (2021). Kuwait's marine biodiversity: qualitative assessment of indicator habitats and species. *Marine Pollution Bulletin*, 163:111915.

Efrem Garedew, Sandewall, M., Söderberg, U., and Campbell, B. M. (2009). Land-Use and Land-Cover Dynamics in the Central Rift Valley of Ethiopia. Environmental Management, 44(4), 683–694.

Egoh, B., Reyers, B., Rouget, M., Richardson, D.M., Le Maitre, D.C., and van Jaarsveld, A.S. (2008) Mapping ecosystem services for planning and management. Agriculture, Ecosystems & Environment 127, 135-140.

Ehrlich, P.R., and Ehrlich, A.H., (1981). Extinction: the causes and consequences of the disappearance of species. Random House, New York.

Eigenbrod, F. Armsworth, PR. Anderson, BJ. Heinemeyer, A. Gillings, S. Roy, DB. Thomas, CD. Gaston, KJ (2010). The impact of proxy-based methods on mapping the distribution of ecosystem services. J. Appl. Ecol. Vol, 47, 377−385.

Elbeltagi, A.; Aslam, M.R.; Malik, A.; Mehdinejadiani, B.; Srivastava, A.; Bhatia, A.S.; Deng, J. (2020). The impact of climate changes on the water footprint of wheat and maize production in the Nile Delta, Egypt. Sci. Total Environ. 743, 140770.

Ellis, C, Erle (2015). Ecology in an anthropogenic biosphere. Ecological Monographs, 85(3), 2015, pp. 287–331.

Elmqvist, T.; Gómez-Baggethun, E. and Langemeyer, J (2016). Ecosystem Services provided by urban green infrastructure. In Routledge Handbook of Ecosystem Services; Potschin, M., Haines-Young, R., Fish, R., Turner, R.K., Eds.; Routledge: New York, NY, USA, pp. 452–464.

Emiru Birhane, Haregeweini Ashfare, Ayele Almaw, Hadgu Hishe, M.A. Gebremedhin, Hailemariam Gebrewahed, Negasi Solomon (2019). LULC changes along topographic gradients in Hugumburda national forest priority area, Northern Ethiopia. Remote Sensing Applications: Society and Environment, 13: 61–68.

Ennaanay, Driss (2006). Impacts of Land Use Changes on the Hydrologic Regime in the Minnesota River Basin. Ph.D. thesis, graduate School, University of Minnesota.

Ermias Teferi, Woldeamlak Bewket and Belay Simane (2016). Effects of land use and land cover on selected soil quality indicators in the headwater area of the Blue Nile basin of Ethiopia. Environmental Monitoring and Assessment 188: 83.

Erskine, P. D., Lamb, D., and Bristow, M. (2006). Tree species diversity and ecosystem function: Can tropical multi-species plantations generate greater productivity? *Forest Ecology and Management, 233(2-3), 205–210.*

Ethiopian Horticulture Development Agency (EHDA) (2012). Ethiopian horticulture sector statistical bulletin, Issue 01. Ethiopian Horticulture Development Agency (EHDA), Addis Ababa, Ethiopia.

Eyasu Elias, Weldemariam Seifu, Bereket Tesfaye and Wondwosen Girmay (2019). Impact of land use cover changes on Lake Ecosystem of Ethiopia central rift valley. Cogent Food and Agriculture, 5: 1595876.

Ezana Haddis Weldeghebrael (2021). Addis Ababa: City Scoping Study. The University of Manchester.

Fadon E., S. Herrera, B. I. Guerrero, M. E. Guerra, and R. Javier, (2020). Chilling and heat requirements of temperate stone fruit trees (prunus sp.)," *Agronomy*, vol. 10, p. 409.

Faleiro F V., and Loyola RD (2013). Socioeconomic and political trade-offs in biodiversity conservation: a case study of the Cerrado Biodiversity Hotspot, Brazil. Divers Distrib. 19(8):977–87.

Fan, F., Weng, Q and Wang, Y (2007). Land use land cover change in Guangzhou, China from 1998 to 2003 based on landsat TM/ETM+ imagery. Sensors 7:1323-1342.

Fargione, J., Hill, J., Tilman, D., Polasky, S and Hawthorne, P (2008). Land clearing and the biofuel carbon debt. Science 319: 1235–1238.

Farley, K. A., Jobbagy, E. G., and Jackson, R. B. (2005). Effects of afforestation on water yield: a global synthesis with implications for policy. *Global Change Biology*, *11(10), 1565–1576.*

Farley, J and Costanza, R. (2010). Payments for ecosystem services: From local to global, Ecological Economics, Vol 69, pp., 2060-2068.

Farm Africa (2022). Making Forests Sustainable: Lessons Learnt from the Bale Eco-Region REDD+ Phase II Project, Addis Ababa, Ethiopia.

Fasika Alemayehu., Motuma Tolera, and Gizaw Tesfaye, (2019). Land use land cover change trend and its drivers in Somodo watershed southwestern, Ethiopia. Afr. J. Agric. Res. 14 (2), 102–117.

Federal Democratic of Ethiopia (FDRE) (2012). Ethiopia's Green Economy Strategy. Addis Ababa, Ethiopia.

Federal Democratic Republic of Ethiopia (FDRE) (2019). Ethiopia's Climate Resilient Green Economy National Adaptation Plan, Addis Ababa, Ethiopia.

Federal Negarit Gazette (2020). Federal Negarit Gazette of the Federal Democratic Republic of Ethiopia Development, Management and Utilization of Community Watersheds Proclamation No.1223/2020 Addis Ababa, Ethiopia.

Federal Negarit Gazette (2021). Federal Negarit Gazette of the Federal Democratic Republic of Ethiopia Capital Market Proclamation; Proclamation No.1248/2021 Addis Ababa, Ethiopia.

Feld, C.K., Da Silva, P.M., Sousa, J.P., De Bello, F., Bugter, R., Grandin, U, Hering, D., Lavorel, S., Mountford, O., Pardo, I., Partel, M., Rombke, J., Sandin, Leonard, Jones, K. Bruce, Harrison, P.(2009). Indicators of biodiversity and ecosystem services: A synthesis across ecosystems and spatial scales. Oikos 118: 1862-1871.

Feng, D., Wu, W., Liang, L., Li, L., and Zhao, G. (2018). Payments for watershed ecosystem services: mechanism, progress and challenges. Ecosystem Health and Sustainability, 4(1), 13–28.

Ferrari, G.T., Ozaki, V., (2014). Missing data imputation of climate datasets: implications to modeling extreme drought events. Revista Brasileira de Meteorologia 29, 21–28.

Fetena, S., Shara, S., Anjulo, A., Gulie, G., Woldesenbet, F., Yilma, B., (2014). Survey on apple production and variety identification in Chencha District of Gamo Gofa Zone, Southern Ethiopia. J. Agric. Food Technol. 4 (5), 7–15.

Fisher, B., Kulindwa, K., Mwanyoka, I., Turner, R. K., & Burgess, N. D. (2010). Common pool resource management and PES: Lessons and constraints for water PES in Tanzania. Ecological Economics, 69(6), 1253–1261.

Fisher, B., Turner, R.K and Morling, P. (2010). Defining and classifying ecosystem services for decision making. Ecological Economics 68, 643–653.

Food and Agriculture Organization (FAO) (1976). A framework for land evaluation. Rome, Italy.

Food and Agriculture Organization (FAO) (2007b). Land evaluation, towards a revised framework. Land and Water Discussion Paper 6. Rome, Italy.

Food and Agriculture Organization (FAO) (2015). Global Forest Resources Assessment 2015: How are the world's forests changing? Second edition. FA. FAO, Rome.54 pp.

Food and Agriculture Organization (FAO) (2016). Payments for forest environmental services in sub-Saharan Africa: A practical guide, by Nantongo, P. Accra, Ghana.

Foody, G. M. (2002). Status of land cover classification accuracy assessment. Remote Sensing of Environment, 80(1), 185–201.

Foster, J (2014). Hiding in plain view: Vacancy and prospect in Paris' Petite Ceinture. Cities 40; 124–132.

Frank, S., Fürst, C., Koschke, L., Witt, A., Makeschin, F., (2013). Assessment of landscape esthetics - validation of a landscape metrics- based assessment by visual estimation of the scenic beauty. Ecol.Indic.32: 222–231.

Fripp E. (2014). Payments for Ecosystem Services (PES): A practical guide to assessing the feasibility of PES projects. Bogor, Indonesia: CIFOR.

Fu, B., Wang, S., Su, C., Forsius, M., (2013). Linking ecosystem processes and ecosystem services. Current Opinion in Environmental Sustainability 5 (1):4–10.

Gebregziabher, G.; Abera, D. A.; Gebresamuel, G.; Giordano, M.; Langan, S. (2016). An assessment of integrated watershed management in Ethiopia. Colombo, Sri Lanka: International Water Management Institute (IWMI). 28p. (IWMI Working Paper 170). doi: 10.5337/2016.214.

Gebreegziabher, T., Suryabhagavan, K. V., and Kumar Raghuvanshi, T. (2021). WebGIS-based decision support system for soil erosion assessment in Legedadi watershed, Oromia Region, Ethiopia. *Geology, Ecology, and Landscapes, 1–18.*

Gebrekidan Worku Tefera, Yihun Taddele Dile, Raghavan Srinivasan, Tracy Baker, Ram L. Ray (2023). Hydrological modeling and scenario analysis for water supply and water demand assessment of Addis Ababa City, Ethiopia. Journal of Hydrology: Regional Studies 46: 101341.

Gebreselassie, S. Hagos, F. Haileslassie, A. Awulachew, S.B. Peden, D. Tafesse, T. (2009). Determinants of adoption of improved land and water management practices in the Blue Nile Basin: exploring strategies for out scaling of promising technologies. Mimeo.

Gebreselassie, S. Oliver K. Kirui and Alisher Mirzabaev. (2016). Economics of Land Degradation and Improvement in Ethiopia, (eds.) Ephraim Nkonya · Alisher Mirzabaev and Joachim von Braun, In: Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development, Springer International Publishing AG Switzerland.

Geijzendorffer, I. R., Martín-López, B., & Roche, P. K. (2015). Improving the identification of mismatches in ecosystem services assessments. *Ecological Indicators, 52, 320–331.*

Geremew Adamu (2013). Assessing the impacts of LULC change on the hydrology of watershed: A case study on Gigel-Abbay Watershed. Lake Tana Basin, Ethiopia. MSc thesis.

Getachew Beyene, Nigussie Dechassa, Alemayehu Regasa, and Lemma Wogi (2022). Land Suitability Assessment for Apple (*Malus domestica*) Production in Sentele Watershed in Hadiya Zone, Southern Ethiopia. Hindawi Applied and Environmental Soil Science.

Getachew Tadesse., E, Zavaleta., C Shennan., and M, FitzSimmons (2014b). Prospect of forest-based ecosystem service in forest-coffee mosaics as forest loss continues in southwestern Ethiopia. Appl Geogr. 50:144–151.

Getahun Kebede (2014). Organic Matter Removal to Reduce Disinfection By-products Precursors; A Case study in "Legedadi Water Treatment Plant. MSc Thesis, Addis Ababa University Institute of Technology, Addis Ababa.

Gete Zeleke and H. Hurni. (2001). Implications of land use/cover changes for mountain resource degradation in the north-western Ethiopian highlands. Mountain Research and Development, 21: 184-191.

Ghorbanzadeh, O., Pourmoradian, S., Blaschke, T., and Feizizadeh, B. (2019). Mapping potential nature-based tourism areas by applying GIS-decision making systems in East Azerbaijan Province, Iran. Journal of Ecotourism, 1–23.

Girma, R., Moges, A., and Quraishi, S. (2015). GIS based physical land suitability evaluation for crop production in Eastern Ethiopia: A case study in Jello Watershed. Agrotechnology, 5(1), 1–7.

Girmay Gebrerufael, Mesfin Menza, Melkamu Mada and Tora Abebe (2014). Empirical Study on Apple Production, Marketing and its Contribution to Household Income in

Chencha District of Southern Ethiopia. Scholarly Journal of Agricultural Science Vol. 4(3), pp. 166-175.

Girmay Kassa (2003). GIS-based analysis of land use/land cover, land degradation and population changes: A study of Boru Metero area of south Wello, Amhara Region, MA Thesis, Department of Geography, Addis Ababa University.

Girvan MS, Bullimore J, Pretty JN, Osborn AM, Ball AS (2003). Soil type is the primary determinant of the composition of the total and active bacterial communities in arable soils. Appl Environ Microbiol 69:1800–1809.

Goldman RL, Tallis H, Kareiva P, and Daily GC (2008). Field evidence that ecosystem service projects support biodiversity and diversify options. Proc Natl Acad Sci U S A. 105(27):9445–8.

Goldman RL and Tallis H (2009). A critical analysis of ecosystem services as a tool in conservation projects: the possible perils, the promises and the partnerships. Ann N Y Acad Sci. 1162:63–78.

Goyal, M.K., and Khan, M (2017). Assessment of spatially explicit annual water-balance model for Sutlej River Basin in eastern Himalayas and Tungabhadra River Basin in peninsular India. Hydrol. Res. 48, 542–558.

Greiber, Thomas (Ed) (2009). Payments for Ecosystem Services. Legal and Institutional Frameworks. IUCN, Gland, Switzerland. xvi + 296 pp.

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M. (2008). Global Change and the Ecology of Cities. Science, 319(5864), 756–760. doi:10.1126/science.1150195.

Guodong Yin., Xiao Wang., Xuan Zhang., Yongshuo Fu., FanghuaHao., and Qiuhong Hu (2020). InVEST Model-Based Estimation of Water Yield in North China and Its Sensitivities to Climate Variables Water 12, 1692.

Habtamu Bewket (2017). Simulating Hydrological and Climate Variability-Impacts on the Watershed of Legedadi Reservoir using ArcSWAT. MSc dissertation Addis Ababa University, Addis Ababa.

Haddad, N.M., Brudvig, L.A., Clobert, J., Davies, K.F., Gonzalez, A., Holt, R.D., Lovejoy, T.E., Sexton, J.O., Austin, M.P., and Collins, C.D., (2015). Habitat fragmentation and its lasting impact on Earth's ecosystems. Science Advances 1, e1500052.

Hagos, F.; van Rooijen, D.; Haileslassie, A.; Yehualashet, H.; Indries, H. (2018). Investigation of the modalities for an innovative financing mechanism for participatory natural resource management in the Bale Eco-region, Ethiopia. Colombo, Sri Lanka: International Water Management Institute (IWMI). 36p. (IWMI Working Paper 181).

Hagos, F. (2012). Impact of Land Registration and Certification on Investment: The Case of Tigray, Northern Ethiopia. Ethiopian Journal of Economics, Vol. XXI, No. 1: 19-47.

Hagos, F., Haileselassie, A., Awulachew, S. B., Mapdeza, E., and Tafesse, T. (2011). Land and water Institutions in the Blue Nile Basin: Setups and gaps for improved land and water management. Review of Policy Research, 28 (2): 149-170.

Hailu, A. H., Kibret, K., and Gebrekidan, H. (2015). Land suitability evaluation for rain fed production of barley and wheat at Kabe Subwatershed, Northeastern Ethiopia. African Journal of Soil Science, ISSN2375-088X, 3(7), 147–156.

Hailu, A., Mammo, S., and Kidane, M. (2020). Dynamics of land use, land cover change trend and its drivers in Jimma Geneti District, Western Ethiopia. *Land Use Policy, 99, 105011*.

Hailu Shiferaw, Woldeamlak Bewket, Tena Alamirew, Gete Zeleke, Demel Teketay, Ketema Bekele, Urs Schaffner, and Sandra Eckert (2019). Implications of land use/land cover dynamics and Prosopis invasion on ecosystem service values in Afar Region, Ethiopia. Sci Total Environ 675:354–366.

Hailu Shiferaw, Tena Alamirew, Tibebu Kassawmar and Gete Zeleke (2021). Evaluating ecosystems services values due to land use transformation in the Gojeb watershed, Southwest Ethiopia. *Environ Syst Res; 10:22.*

Hall, L.S.; Krausman, P.R.; Morrison, M.L (1997). The habitat concept and a plea for standard terminology. Wildl. Soc. Bull. 25, 171–182.

Hamere Yohannes and Teshome Soromessa (2018). Land suitability assessment for major crops by using GIS-based multi-criteria approach in Andit Tid watershed, Ethiopia, Cogent Food & Agriculture, 4:1, 1470481.

Hamere Yohannes, Teshome Soromessa, Mekuria Argaw and Ashraf Dewan (2021). Spatio-temporal change in habitat quality and linkage with landscape characteristic in the Beressa watershed. Journal of Environmental Management, 281:11188.

Hana Yeshaw (2020). Farmer's Perception and Factors Affecting Watershed Management Practice: A Case of Legedadi Reservoir Catchment Area. MA thesis, Addis Ababa University, Addis Ababa.

Handoro F and Gemu M (2007). A survey of temperate fruit diseases in SNNPR, Ethiopia. African crop sciences conference proceedings, Egypt.

Harrison, P. A., Berry, P. M., Simpson, G., Haslett, J. R., Blicharska, M., Bucur, M., and Turkelboom, F. (2014). Linkages between biodiversity attributes and ecosystem services: A systematic review. Ecosystem Services, 9, 191–203.

Havlicek, E., Mitchell, E.A.D. (2014). Soils Supporting Biodiversity. In: Dighton, J., Krumins, J. (eds) Interactions in Soil: Promoting Plant Growth. Biodiversity, Community and Ecosystems, vol 1. Springer, Dordrecht.

Hayesso T (2008). Hope in apples. SNV Netherlands Development Organisation: Case studies 2008.

Hegazy, I. R., and Kaloop, M. R. (2015). Monitoring urban growth and land use change detection with GIS and remote sensing techniques in Daqahlia governorate Egypt. International Journal of Sustainable Built Environment, 4(1), 117–124.

Hegetschweiler, K. T., de Vries, S., Arnberger, A., Bell, S., Brennan, M., Siter, N., … Hunziker, M. (2017). Linking demand and supply factors in identifying cultural ecosystem services of urban green infrastructures: A review of European studies. Urban Forestry & Urban Greening, 21, 48–59.

Helnata Tilahun (2019). Key drivers of water quality in Dire and Legedadi reservoirs, Ethiopia. MSc thesis Addis Ababa University, Addis Ababa.

Hepburn L, Smith AC, Zelenski J, Fahrig L (2021). Bird Diversity Unconsciously Increases People's Satisfaction with Where They Live. *Land*. 10(2):153.

Hernández-Morcillo, M., Plieninger, T., and Bieling, C. (2013). An empirical review of cultural ecosystem service indicators. *Ecological Indicators, 29, 434–444.*

Herold, M., Goldstein, N. C., and Clarke, K. C. (2003). The spatiotemporal form of urban growth: Measurement, analysis and modeling. Remote Sensing of Environment, 86(3), 286–302.

Hester S. M. and Cacho, O (2003). Modelling apple orchard systems," *Agricultural Systems*, vol. 77, no. 2, pp. 137–154.

Hill, GW, PR Courtney (2006). Demand analysis projections for recreational visits to countryside woodlands in Great Britain. Forestry 79: 18-200.

Holden, S. T., Deininger, K., and Ghebru, H. (2009). Impacts of low-cost land certification on investment and productivity. American Journal of Agricultural Economics, 91(2), 359–373.

Hooper D. U., Chapin F. S., J. J. Ewel, A. Hector, P. Inchausti, S. Lavorel, J. H.Lawton, D. M. Lodge, M. Loreau, S. Naeem, B. Schmid, H. Setälä, A. J. Symstad, J.Vandermeer and D. A. Wardle (2005). Effects of Biodiversity on Ecosystem Functioning: A Consensus of Current Knowledge Ecological Monographs, Vol. 75, No. 1 pp. 3-35.

Houghton, R. A., Hackler J. L., and Lawrence K. T. (1999). The U.S. carbon budget: contributions from land-use change, Science, 285:574–578.

Hu, W.; Li, G.; Gao, Z.; Jia, G.; Wang, Z.; Li, Y (2020). Assessment of the impact of the Poplar Ecological Retreat Project on water conservation in the Dongting Lake wetland region using the InVEST model. Sci. Total Environ. 733, 139423.

Huabin, H, Liu, W.; Min C (2008). Impact of land use and land cover changes on ecosystem services in Menglun, Xishuangbanna, Southwest China. Environ. Monit. Assess., 146: 47−156.

Ib Friis, SebsebeDemissew and Paulo van Breugel (2010). Atlas of the potential vegetation of Ethiopia. The Royal Danish Academy of Sciences and Letters, Copenhagen, Denmark.

ICOLD (2009). Sedimentation and Sustainable Use of Reservoirs and River Systems, Bulletin.

International Panel on Climate Change (IPCC) (2000). IPCC Special Report on LandUse, Land Use Change and Forestry. Summary for Policy Makers. Geneva, Switzerland:

International Panel on Climate Change (IPCC) (2019). IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse gas fluxes in Terrestrial Ecosystems. Intergovernmental Panel on Climate Change.

Isbell, F., Calcagno, V., Hector, A., Connolly, J, Stanley Harpole W., Reich, P., Scherer-Lorenzen, M, Schmid, B. , Tilman,D,. Ruijven, J,. Weigelt, A. Wilsey, B., Zavaleta, E and Loreau, M (2011). High plant diversity is needed to maintain ecosystem services. *Nature* **477:** 199–202.

Jack, B. K., Kousky, C., and Sims, K. R. E. (2008). Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms. Proceedings of the National Academy of Sciences, 105(28), 9465–9470.

Jan Nyssen, Jean Poesen, Sil Lanckriet, Miro Jacob, Jan Moeyersons, Mitiku Haile, Nigussie Haregeweyn, R. Neil Munro, Katrien Descheemaeker, Enyew Adgo, Amaury Frankl, and Jozef Deckers (2015). Land Degradation in the Ethiopian Highlands. World Geomorphological Landscapes.

Jeetendra Kumar Patel and Raj Bahadur Anuragi (2023). Identifying the Ecotourism Potential Locations using AHP and Weighted Overlay Analysis: A case of Bundelkhand region in Central India, Papers in Applied Geography, DOI: 10.1080/23754931.2023.2254786

Jensen, J.R (2000). Remote Sensing of the Environment: An Earth Resource Perspective; Prentice Hall: Upper Saddle River, NJ, USA, p. 544.

Jie, G.; Feng, L.; Hui, G.; Zhou, C and Zhang, X (2015). The impact of land-use changes on water-related ecosystem services: A study of the Guishui River Basin, Beijing, China. J. Clean. Prod. 163, S148–S155.

Johnson MD (2007). Measuring habitat quality: a review. The Condor. 109:489–504.

Jones M, S., Luisetti, T., Turner, R.K., Fisher, B. (2011). Ecosystem valuation: some principles and a partial application. Environmetrics, 22(5): 675–685.

Joram K. Kagombe, Joshua K. Cheboiwo, Alfred Gichu, Collins Handa and Jane Wamboi (2018).Payment for Environmental Services: Status and Opportunities in Kenya. Journal of Resources Development and Management. Volume 40: ISSN 2422-8397.

Jose S (2009). Agroforestry for ecosystem services and environmental benefits: anoverview. Agroforestry Systems, 76:1-10.

Jose, S. and Bardhan, S (2012). Agroforestry for Biomass Production and Carbon Sequestration: An Overview. Agroforest Syst. 86:105–111.

Jyotish Prakash Basu (2013). Agroforestry, climate change mitigation and livelihood security in India. Journal of Forestry Science, 44(Suppl 1): S11.

Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S. (2011). Natural Capital: Theory and Practice of Mapping Ecosystem Services. Oxford University Press, New York, USA, pp. 432.

Karma Chozom and Gibji Nimasow (2021). GIS- and AHP-based land suitability analysis of Malus domestica Borkh. (apple) in West Kameng district of Arunachal Pradesh, India. Applied Geomatics.

Kasperson, J.X., Kasperson, R.E. and Turner, B.L. II (1995). Regions at Risk: Comparisons of Threatened Environments. United Nations Univ. Press, Tokyo.

Kazakeviciute-Januskeviciene, G., Janusonis, E., Bausys, R., Limba, T., & Kiskis, M. (2020). Assessment of the Segmentation of RGB Remote Sensing Images: A Subjective Approach. Remote Sensing, 12(24), 4152.

Ken Norris (2012). Biodiversity in the context of ecosystem services: the applied need for systems approaches. Phil. Trans. R. Soc. B (2012) 367, 191–199.

Kerr, J. Milne, G. Chhotray, V. Baumann, P. and James, A.J. (2007). Managing watershed externalities in India: theory and practice, Environment, Development and Sustainability, 9:263–281.

Kifle Arsiso, B., Mengistu Tsidu, G., Stoffberg, G. H., and Tadesse, T. (2017). Climate change and population growth impacts on surface water supply and demand of Addis Ababa, Ethiopia. Climate Risk Management, *18:21–33.*

Kim H, and Shim K (2018). Land suitability assessment for apple (*Malus domestica*) in the Republic of Korea using integrated soil and climate information, MLCM, and AHP. Int J Agric & Biol Eng, 11(2): 139–144.

Kindu Mengiste, Schneider T, Demel Teketay, and Knoke T. (2016). Change of ecosystem service values in response to land use/ cover dynamic in Munessa–Shashemene landscape of the Ethiopian highlands. Sci Total Environ. 547:137–147.

Kissinger G, Patterson C, Neufeldt H. (2013). Payments for ecosystem services schemes: project-level insights on benefits for ecosystems and the rural poor. ICRAF Working Paper No 172, Nairobi, Kenya.

Kleijn D, Kohler F, Báldi A, Batáry P, Concepción E.D, Clough Y, Díaz M, Gabriel D, Holzschuh A, Knop E, Kovács A, Marshall E.J.P, Tscharntke T and Verhulst J (2009). On the relationship between farmland biodiversity and land-use intensity in Europe. Proc. R. Soc. B.276:903–909.

Klein, A. M., Vaissiere, B. E., Cane, J. H., SteffanDewenter, I., Cunningham, S. A., Kremen, C. and Tscharntke, T. (2007). Importance of pollinators in changing landscapes for world crops. Proc. R. Soc. B, 274; 303 –313.

Knight AT, Cowling RM, and Campbell BM (2006). An operational model for implementing conservation action. Conserv Biol. 20(2):408–19. PMID: 16903102 21.

Knoke, T., Steinbeis, O.-E., Bösch, M., Román-Cuesta, R.M., and Burkhardt, T., (2011). Cost-effective compensation to avoid carbon emissions from forest loss: an approach to consider price–quantity effects and risk-aversion. Ecol. Econ. 70, 1139–1153.

Kordi, M., and Brandt, A. (2012). Effects of increasing fuzziness on analytic hierarchy process for spatial multicriteria decision analysis. Computers, Environment and Urban Systems, 36, 43e53.

Kremen Claire (2005). Managing ecosystem services: what do we need to know about their ecology? Ecology Letters, (2005) 8: 468–479.

Kreuter U.P., Harris H.G., Matlock M.D., Lacey R.E. Change in ecosystem service values in the San Antonio area, Texas. Ecol. Econ. 2001;39(3):333–346.

Krishna K. R. (2014). Agroecosystems, Soils, Climate, Crops, Nutrient Dynamics, and Productivity, CRC Press, Boca Raton, FL, USA.

Lakew Desta, Carucci, V., Asrat Wendem-Ageňehu and Yitayew Abebe (eds). (2005). Community Based Participatory Watershed Development: A Guideline. Ministry of Agriculture and Rural Development, Addis Ababa, Ethiopia

Lakiso A. N. (1980). Correlation of fisheye photography to canopy structure, light climate, and biological response to light in apple trees, *Journal of the American Society for Horticultural Science*, vol. 105, pp. 43–46.

Lang, Y.; Song, W.; Zhang, Y (2017). Responses of the water-yield ecosystem service to climate and land use change in Sancha River Basin, China. Phys. Chem. Earth Parts A/B/C, 101, 102–111.

Langat D., Otuoma J., Kagombe J., Cheboiwo J.K., Ongugo P. and Kigomo B. (2017). Guidelines for Establishing Payment for Ecosystem Services Schemes in Kenya. KEFRI, Nairobi.

Laurance, W. F., Lovejoy, T. E and Vasconcelos, H. L (2002). Ecosystem decay of Amazonian forest fragments: A 22-year investigation. Conservation Biology16:605–618.

Lawler, J.J., Lewis, D.J., Nelson, E., Plantinga, A.J., Polasky, S., Withey, J.C., Helmers, D. P., Martinuzzi, S., Pennington, D., Radeloff, V.C., (2014). Projected land-use change impacts on ecosystem services in the United States. Proc. Natl. Acad. Sci. U.S.A. 111 (20), 7492–7497.

Legesse N., (2010). Ethiopia's Indigenous Tees biology, uses and propagation techniques. Addis Ababa University Press. Addis Ababa, Ethiopia.

Lemlem Tajebe, Agena Anjulo and Zeleke Ewnetu (2013). Apple Based Agroforestry in Dendi Woreda, Oromiya Region: Income Contribution and Determinants for Adoption. Ethiop. J. Agric. Sci. 23:61-73.

Li, R.Q.; Dong, M.; Cui, J.Y.; Zhang, L.L.; Cui, Q.G.; He, W.M (2007). Quantification of the Impact of Land-Use Changes on Ecosystem Services: A Case Study in Pingbian County, China. Environ. Monit. Assess., 128, 503−510.

Li, X.; Hou, X.; Song, Y.; Shan, K.; Zhu, S.; Yu, X. and Mo, X (2018). Assessing Habitat Quality Changes for Shorebirds in Stopover Sites: A Case Study in Yellow River Delta, China. Wetlands, 39, 67–77.

Liambila, R. N., and Kibret, K. (2016). Climate change impact on land suitability for rain fed crop production in Lake Haramaya watershed, eastern Ethiopia. Journal of Earth Science & Climatic Change, 7(3), 1–12.

Lian, X.-H.; Qi, Y.; Wang, H.-W.; Zhang, J.-L.; Yang, R (2020). Assessing Changes of Water Yield in Qinghai Lake Watershed of China. Water 12:11.

Lillesand, T., Kiefer, R.W., Chipman, J., (2015). Remote Sensing and Image Interpretation. John Wiley & Sons.

Lipper, L., Thornton, P., Campbell, B.M., Baedeker, T., Braimoh, A., Bwalya, M., Caron, P., Cattaneo, A., Garrity, D., Henry, K., Hottle, R., Jackson, L., Jarvis, A., Kossam, F., Mann, W., McCarthy, N., Meybeck, A., Neufeldt, H., Remington, T., Sen, P.T., Sessa, R., Shula, R., Tibu, A., Torquebiau, E.F., (2014). Climate-smart agriculture for food security. Nat. Clim. Chang. 4, 1068–1071.

Liting Xu, Sophia Shuang Chen, Yu Xu, Guangyu Li, and Weizhong Su (2019). Impact of Land-Use alteration on Habitat quality between 1985–2015 in the Taihu Lake Basin. Sustainability, 11:3513.

Liu, C.F.; Wang, C.; Liu, C.L (2018). Spatio-temporal variation on habitat quality and its mechanism within the transitional area of the Three Natural Zones: A case study in Yuzhong county. Geogr. Res. 37, 419–432.

Liu, H., Remme, R. P., Hamel, P., Nong, H., and Ren, H. (2020). Supply and demand assessment of urban recreation service and its implication for greenspace planning-A case study on Guangzhou. *Landscape and Urban Planning, 203, 103898.*

Loomis, JB, and Richardson RB (2006). An external validity test of intended behavior: comparing revealed preference and intended visitation in response to climate change. Journal of Environmental Planning and Management 49: 621-630.

Lorena Peña, Izaskun Casado-Arzuaga., and Miren Onaindia (2015). Mapping recreation supply and demand using an ecological and a social evaluation approach Ecosystem Services 13:108–118.

Lorenz, K., and Lal, R., (2014). Soil organic carbon sequestration in agroforestry systems. A review. Agron. Sustain. Dev. 34, 443–454.

Loureiro, ML, F Macagno, PA Nunes, R Tol (2012). Assessing the impact of biodiversity on tourism flows: an econometric model for tourist behavior with implications for conservation policy. Journal of Environmental Economics and Policy 1: 174-194.

Lu, D and Weng, Q (2007). A survey of image classification methods and techniques for improving classification performance. Int. J. Remote Sens.Vol 28, 823−870.

Lu, H.; Yan, Y.; Zhu, J.; Jin, T.; Liu, G.; Wu, G.; Stringer, L.C.; Dallimer, M (2020). Spatiotemporal water yield variations and influencing factors in the Lhasa River Basin, Tibetan Plateau. Water 12: 1498.

Luck Gary W., Gretchen C. Daily, Paul R. Ehrlich (2003). Population diversity and ecosystem services, Trends in Ecology and Evolution; 18(7):331-336.

Mace, G.M.; Norris, K.; Fitter, A.H (2012). Biodiversity and ecosystem services: A multilayered relationship. Trends Ecol. Evol, 27: 19–26.

Madrigal-Martínez S, Puga-Calderón RJ (2018). Land suitability and sensitivity analysis for planning apple growing in Mala's valley. Peru. Bioagro 30(2):95–106.

Maes J, Egoh B, Willemen L, Liquete C, Vihervaara P, Schägner JP, Bruna Grizzetti , Evangelia G. Drakou , Alessandra La Notte, Grazia Zulian, Faycal Bouraoui, Maria Luisa Paracchini, Leon Braat, Giovanni Bidoglio (2012). Mapping ecosystem services for policy support and decision-making in the European Union. Ecosyst Serv. 1:31–9.

Maes, J., and Jacobs, S., (2017). Nature-based solutions for Europe's sustainable development. Conservation Letters 10, 121-124.

Malczewski, J. (1999). GIS and multi-criteria decision analysis. Wiley: New York.

Malczewski, J (2000). "Local weighted linear combination," *Transactions in GIS*, vol. 15, no.4, pp. 430–455.

Manandhar S, Pandey VP, Kazama F (2014). Assessing suitability of apple cultivation under climate change in mountainous regions of Western Nepal. Reg Environ Change 14:743–756.

Manjaiah, K.M.; Sandeep, S.; Ramesh, T.; Mayadevi, M.R (2017). Soil Organic Carbon Stocks Under Different Agroforestry Systems of North-Eastern Regions of India. In Agroforestry: Anecdotal to Modern Science; Dagar, J.C., Tewari, V.P., Eds.; Springer: Singapore,pp. 299–316.

Mansouri Daneshvar M.R. (2014). Landslide susceptibility zonation using analytical hierarchy process and GIS for the Bojnurd region, northeast of Iran. Landslides 11:1079–1091.

Måren, I. E., BhattaraI, K. R., and Chaudhary, R. P. (2014). Forest ecosystem services and biodiversity in contrasting Himalayan Forest management systems. *Environmental Conservation, 41(01): 73–83.*

Martínez-Harms, M. J., and Balvanera, P. (2012). Methods for mapping ecosystem service supply: a review. *International Journal of Biodiversity Science, Ecosystem Services & Management, 8(1-2), 17–25.*

Mas, J.F., Velazquez, A., Gallegos, J.R.D., Saucedo, R.M., Alcantare, C., Bocco, G., Castro, R., Fernandez, T and Vega, A.P (2004). Assessing land use/cover changes: A nationwide multidate spatial database for Mexico. International journal of applied earth observation and geo-information. 5:249-261.

Mattias Gaglio, Vassilis George Aschonitis, Marta Maria Mancuso, Juan Pablo Reyes Puig, Francisco Moscoso, Giuseppe Castaldelli & Elisa Anna Fano (2017).

Changes in landuse and ecosystem services in tropical forest areas: a case study in Andes mountains of Ecuador, International Journal of Biodiversity Science, Ecosystem Services & Management, 13:1, 264-279.

Mbow, C.; Smith, P.; Skole, D.; Duguma, L.; Bustamante, M. (2014). Achieving Mitigation and Adaptation to Climate Change through Sustainable Agroforestry Practices in Africa. Curr. Opin. Environ. Sustain. 6: 8–14.

Mbow, C. (2015). Agroforestry Can Form an Effective, Efficient and Fair Pathway to Achieve Food Security and Agricultural Sustainability in Africa. Climate Change and Development-ICRAF, World Agroforestry Centre, Nairobi, Kenya.

McAdam, J., Burgess, P., Graves, A., Riguero-Rodríquez A., and Mosquera-Losada, M.R., (2009). Classifications and functions of agroforestry systems in Europe. In: Rigueiro-Rodríguez, A., McAdam, J, Mosquera-Losada, M. (eds) Agroforestry in Europe: Current Status and Future Prospects. Springer Science + Business Media B.V., Dordrecht, pp. 21–42.

McBride MF, Wilson K a, Bode M, Possingham HP (2007). Incorporating the effects of socioeconomic uncertainty into priority setting for conservation investment. Conserv Biol. 21(6):1463–74.

McVittie A, and Hussain S. (2013). The economics of ecosystems and biodiversity-valuation database manual. Edinburgh: Sustainable Ecosystems Team at the Scotland’s Rural College.

Memon A. J and Thapa B.G (2016). Explaining the de facto open access of public property commons: Insights from the Indus Delta mangroves, Environmental Science & Policy, Vol 66, pp. 151-159.

Menale Wondie., Schneider, W., Assefa Melesse and Demel Teketay. (2011). Spatial and Temporal Land Cover Changes in the Simen Mountains National Park, a World Heritage Site in Northwestern Ethiopia. Remote Sensing, 3, 752-766.

Mertz Ole, Helle M R, Gabor L. Lövei, Ivan Nielsen, Cecil C and Konijnendijk (2007). Ecosystem services and biodiversity in developing countries. Biodivers Conserv, 16:2729–2737.

Mesfin Sahle and Kumelachew Yeshitela (2018). Dynamics of land use land cover and their drivers study for management of ecosystems in the socio-ecological landscape of Gurage Mountains, Ethiopia, Remote Sensing Applications: Society and Environment.

Mesfin Sahle., Osamu Saito., Christine Fürst., and Kumelachew Yeshitela (2019). Quantifying and mapping of water-related ecosystem services for enhancing the security of the food-water-energy nexus in tropical data-sparse catchment. Science of the Total Environment 646:573–586.

Millennium Ecosystem Assessment (MEA) (2005). Ecosystems and Human Well-being: Synthesis. Island Press, Washington, DC.

Mills, AS, TN Westover (1987). Structural differentiation: a determinant of park popularity. Annals of Tourism Research 14: 486-498.

Mo, W.; Zhao, Y.; Yang, N.; Xu, Z.; Zhao,W.; Li, F (2021). Effects of Climate and Land Use/Land Cover Changes onWater Yield Services in the Dongjiang Lake Basin.ISPRS Int.J.Geo-Inf. 10, 466.

Mohammad-Taghi Sattari, Ali Rezazadeh-Joudi and Andrew Kusiak (2017). Assessment of different methods for estimation of missing data in precipitation studies. Hydrology Research 48 (4): 1032–1044.

Morales-Quintana L., Waite, J. M. Kalcsits, L. Torres, C. A. and Ramos P. (2020). Sun injury on apple fruit: physiological, biochemical and molecular advances, and future challenges, *Scientia Horticulturae*, vol. 260, Article ID 108866.

Motuma, M., Suryabhagavan, K. V., and Balakrishnan, M. (2016). Land suitability analysis for wheat and sorghum crops in Wogdie District, South Wollo, Ethiopia, using geospatial tools. Applied Geomatics, 8(1), 57–66.

Muleta, S., Yohannes, F., and Rashid, S. M. (2006). Soil erosion assessment of Lake Alemaya watershed. Journal of Land Degradation & Development, 17,333-341.

Murtinho, F., and Hayes, T (2017). Communal Participation in Payment for Environmental Services (PES): Unpacking the Collective Decision to Enroll. *Environmental Management* **59,** 939–955.

Murthy IK, Gupta M, Tomar S, Munsi M, Tiwari R, Hegde GT, Ravindranath NH (2013). Carbon sequestration potential of agroforestry systems in India. Journal of Earth Science & Climate Change 4(1):1-7.

Mustafa, A. A., Man Singh1, R. N Sahoo, Nayan Ahmed, Manoj Khanna, A. Sarangi and A. K. Mishra (2011). Land suitability analysis for different crops: a multi-criterion decision-making approach using remote sensing and GIS, *Researcher*, vol. 3, no. 12, pp. 61–84.

Muzein, B. S. (2006). Remote sensing and GIS for land cover, land use change detection, and analysis in the semi-natural ecosystems and agriculture landscapes of the central Ethiopian Rift Valley. Verlag nicht ermittelbar.

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GAB, Kent J (2000). Biodiversity hotspots for conservation priorities. Nature. 403:853–8. PMID: 10706275 10.

Nahuelhual, L., Carmona, A., Lozada, P., Jaramillo, A., Aguayo, M. (2013). Mapping recreation and ecotourism as a cultural ecosystem service: an application at the local level in Southern Chile.Appl.Geogr.40:71–82.

Nahusenay, A., and Kibebew, K. (2015). Land suitability evaluation in Wadla Delanta Massif of north central highlands of Ethiopia for rain fed crop production. African Journal of Agricultural Research, 10(13), 1595–1611.

Naidoo R, Balmford A, Ferraro PJ, Polasky S, Ricketts TH, Rouget M (2006). Integrating economic costs into conservation planning. Trends Ecol Evol. 21(12):681–7. PMID: 17050033 11.

Nair, P.K.R., Nair, V.D., Kumar, B.M., Showalter, J.M., (2010). Carbon sequestration in agroforestry systems, in: Advances in Agronomy. pp. 237–307.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., and Shaw, M. (2009). Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7: 4-11.

Nepal S., Flügel WA., and Shrestha A.B. (2014). Upstream-downstream linkages of hydrological processes in the Himalayan region. Ecological Processes 3(19):1– 16.

Neuvonen, M, E Pouta, J Puustinen, T Sievänen (2010). Visits to national parks: effects of park characteristics and spatial demand. Journal for Nature Conservation 18: 224-229.

Nie, W.; Yuan, Y.; Kepner, W.; Maliha, S.; Michael, J.; Caroline, E (2011). Assessing impacts of land use and land cover changes on hydrology for the upper San Pedro watershed. J. Hydrol. 407, 105–114.

Nigussie Senait, Li Liu, and Yeshitela Kumelachew (2021). Indicator development for assessing recreational ecosystem service capacity of urban green spaces - A participatory approach. Ecological Indicators, 121: 107026.

Niklas Möhring and Robert Finger (2022). Pesticide-free but not organic: Adoption of a large-scale wheat production standard in Switzerland, Food Policy, Volume 106.

Ningrum A, SetiawanY, and TariganS D (2022). Annual Water Yield Analysis with InVEST Model in Tesso Nilo National Park, Riau Province *IOP Conf. Ser.: Earth Environ. Sci.*950 012098**.**

Nino, K., Mamo, Y., Mengesha, G., and Kibret, K. S. (2017). GIS based ecotourism potential assessment in Munessa Shashemene Concession Forest and its surrounding area, Ethiopia. Applied Geography, 82, 48–58.

Norliyana Adnan and Mohd Parid Mamat (2020). Spatial Analysis Model Assessing Habitat Quality of Selangor. IOP Conf. Ser.: Earth Environ. Sci. 549:012049.

Nyong, A. P., Ngankam, T. M., and Felicite, T. L. (2019). Enhancement of resilience to climate variability and change through agroforestry practices in smallholder farming systems in Cameroon. *Agroforestry Systems.*

Nyongesa J. M, Bett H. K., Lagat J. K. and Ayuya O. I. (2016). Estimating farmers' stated willingness to accept pay for ecosystem services: case of Lake Naivasha watershed Payment for Ecosystem Services scheme-Kenya. Ecological Processes, 5:15.

Oelbermann M, Voroney RP, Gordon AM (2004). Carbon sequestration in tropical and temperate agroforestry systems: a review with examples from Costa Rica and southern Canada. Agriculture Ecosystems and Environment, 104:359-377.

Olaf Bastian (2013). The role of biodiversity in supporting ecosystem services in Natura 2000 sites. Ecological Indicators 24: 12–22.

Oliver, T.H., Heard, M.S., Isaac, N.J.B., Roy, D.B., Procter, D., Eigenbrod, F., Freckleton, R., Hector, A., Orme, C.D.L., Petchey, O.L., Proença, V., Raffaelli, D., Suttle, K.B., Mace, G.M., Martín-Lopez, B., Woodcock, B.A., and Bullock, J.M., (2015). Biodiversity and resilience of ecosystem functions. Trends Ecol. Evol, 11: 673-684.

Okiria, E.; Zaki, M.K.; and Noda, K (2021**)**. A Review of Payment for Ecosystem Services (PES) in Agricultural Water: Are PES from the Operation of Agricultural Water Control Structures Ubiquitous? *Sustainability*, *13*, 12624.

Osborne P (2000). Tropical ecosystems and ecological concepts. Cambridge University Press, Cambridge, UK. p. 280-298.

Pande B. Chaitanya., Kanak N. Moharir., Balamurugan Panneerselvam., Sudhir Kumar Singh., Ahmed Elbeltagi., Quoc Bao Pham., Abhay M. Varade., and J. Rajesh (2021a). Delineation of groundwater potential zones for sustainable development and planning using analytical hierarchy process (AHP), and MIF techniques. *Appl Water Sci* 11, 186.

Pande, C. B., and Moharir, K. (2015). GIS based quantitative morphometric analysis and its consequences: a case study from Shanur River Basin, Maharashtra India. Applied Water Science, 7(2), 861–871.

Pande, C. B., Moharir, K. N., Kumar Singh, S., Varade, A. M., Elbeltagi, A., Khadri, S. F. R., and Choudhari, P. (2021). Estimation of crop and forest biomass resources in a semi-arid region using satellite data and GIS. Journal of the Saudi Society of Agricultural Sciences, 20(5), 302–311.

Pande, C. B., Kadam, S. A., Jayaraman, R., Gorantiwar, S., & Shinde, M. (2021). Prediction of soil chemical properties using multispectral satellite images and wavelet transforms methods. Journal of the Saudi Society of Agricultural Sciences.

Paracchini, M. L., Zulian, G., Kooperoinen, L., Schägner, J. P., Termansen, M., Zander- sen, M., Perez-Soba, M., Scholefield, P. A., Bidoglio, G. (2014). Mapping cultural ecosystem services: a framework to assess the potential for outdoor recreation across the EU.Ecol.Indic.45:371–385.

Peijie Wei., Shengyun Chen., Minghui Wu., Yanfang Deng., HaojieXu., YinglanJia., and Fang Liu (2021). Using the InVEST Model to Assess the Impacts of Climate and Land Use Changes on Water Yield in the Upstream Regions of the Shule River Basin. Water; 13:1250.

Pereira, H.M., Reyers, B., Watanabe, M., Bohensky, E., Foale, S., Palm, C., Espaldon, M.V., Armenteras, D., Tapia, M., and Rincon, A., (2005). Condition and trends of ecosystem services and biodiversity. Ecosystems and Human Well-being: Sub-global. Island Press, p. Condition and trends of ecosystem services and biodiversity.

Pessacg, N.; Flaherty, S.; Brandizi, L.; Solman, S.; Pascual, M (2015). Getting water right: A case study in water yield modelling based on precipitation data. Sci. Total Enviorn. 537, 225–234.

Plieninger, T., Dijks, S., Oteros-Rozas, E., & Bieling, C. (2013). Assessing, mapping, and quantifying cultural ecosystem services at community level. Land Use Policy, 33, 118–129.

Pohjonen, V. (1989). Establishment of fuel wood plantations in Ethiopia. University of Joensuu, Finland, 7-387.

Polasky S, Nelson E, Lonsdorf E, Fackler P, Starfield A (2005). Conserving species in a working landscape: land use with biological and economic objectives. Ecol Appl.15(4):1387–401. 15.

Polasky Stephen., Erik Nelson., Derric Pennington., Kris A. Johnson (2011). The Impact of Land-Use Change on Ecosystem Services, Biodiversity and Returns to Landowners: A Case Study in the State of Minnesota. *Environ Resource Econ* **48,** 219–242.

Pompa-García, M.; Solís-Moreno, R.; Rodríguez-Téllez, E.; Pinedo-Álvarez, A.; Avila-Flores, D.;Hernández-Díaz, C.; Velasco-Bautista, E (2015). Viewshed analysis for improving the effectiveness of watchtowers, in the north of Mexico. Open For. Sci. J. 2010, 3, 17–22.

Porras, I., Grieg-Gran, M. and Neves, N. (2008). All that glitters. A review of payments for watershed services in developing countries. Natural Resource Issues No. 11. International Institute for Environment and Development. London, UK.

Potts, S.G., Biesmeijer, J.C and Kremen, C. (2010). Global pollinator declines: Trends, impacts and drivers. Trends in Ecology and Evolution 25: 345–353.

Pramanik MK (2016). Site suitability analysis for agricultural land use of Darjeeling district using AHP and GIS techniques. Model Earth Syst Environ 2:56.

Pueffel, C.; Haase, D.; and Priess, J.A (2018). Mapping ecosystem services on brownfields in Leipzig, Germany. Ecosyst. Serv. 30, 73–85.

Rabia, H Ahmed and Terribile Fabio (2013). Introducing a New Parametric Concept for Land Suitability Assessment. International Journal of Environmental Science and Development; 4:1.

Raffaelli, D. G. (2006). Biodiversity and ecosystem functioning: issues of scale and trophic complexity. Mar. Ecol. Prog. Ser. 311, 285–294.

Rajesh, J., Pande, C.B., Kadam, S.A., Sunil D. Gorantiwar, and Mukund G. Shinde (2021). Exploration of groundwater potential zones using analytical hierarchical process (AHP) approach in the Godavari River basin of Maharashtra in India. *Appl Water Sci* **11**, 182.

Ramirez F. and Kallarackal J. (2014). Ecophysiology of temperate fruit trees in the tropics, *Advances in Environmental Research*, vol. 31.

Randev, A. K. (2009). Impact of climate change on apple productivity in Himachal Pradesh, *60th International Executive Council Meeting and 5th Asian Regional Conference*.

Ranjith P. Udawatta, J. John Krstansky, Gray S. Henderson, and Harold E. Garrett (2002). Agroforestry Practices, Runoff, and Nutrient Loss: A Paired Watershed Comparison. J. Environ. Qual. 31:1214–1225.

Rawat, J., and Kumar, M. (2015). Monitoring land use/cover dynamics using remote sensing and GIS techniques: A case study of Hawalbagh block, district Almora, Uttarakhand, India. The Egyptian Journal of Remote Sensing and Space Science, 18(1), 77–84.

Redhead, J.W.; Stratford, C.; Sharps, K.; Jones, L.; Ziv, G.; Clarke, D.; Oliver, T.H.; Bullock, J.M (2016). Empirical validation of the InVEST water yield ecosystem service model at a national scale. Sci. Total Environ. 569–570, 1418–1426.

Rémi Cardinael, Tiphaine Chevallier, Bernard G. Barthès, Nicolas P.A. Saby, Théophile Parent, Christian Dupraz, Martial Bernoux and Claire Chenu (2015). Impact of alley cropping agroforestry on stocks, forms and spatial distribution of soil organic carbon - a case study in a Mediterranean context. Geoderma, Elsevier, 2015, 259–260, pp.288-299.

Ribeiro MC, Martensen AC, Metzger JP, Tabarelli M, Scarano F, Fortin M-J (2011). The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In: Zachos FE, Habel JC, editors. Biodiversity hotspots: distribution and protection of conservation priority areas. 1st ed. London New York: Springer; p. 405–34.

Richardson, R., and Loomis, J B (2005). Climate change and recreation benefits in an alpine national park. Journal of Leisure Research 37: 307-320.

Robert J. Zomer, Henry Neufeldt, Jianchu Xu, Antje Ahrends, Deborah Bossio, Antonio Trabucco, Meine van Noordwijk, Mingcheng Wang (2016). Global Tree Cover and Biomass Carbon on Agricultural Land: The contribution of agroforestry to global and national carbon budgets. *Scientific Reports*, 6: 29987.

Robertson N, and Wunder S (2005). Fresh tracks in the forest: Assessing incipient payments for environmental services initiatives in Bolivia. Bogor, Indonesia: CIFOR; 152 p.

Rode, J., Wittmer, H., Emerton, L., and Schröter-Schlaack, C. (2016). "Ecosystem service opportunities": A practice-oriented framework for identifying economic instruments to enhance biodiversity and human livelihoods. *Journal for Nature Conservation, 33, 35–47.*

Rodenhouse, N. L., Sillett, T. S., Doran, P. J., and Holmes and R. T. (2003). Multiple density-dependence mechanisms regulate a migratory bird population during the breeding season. Proc. R. Soc. B Biol. Sci. 270, 2105–2110.

Rodríguez-Prieto, I., Bennett, V. J., Zollner, P. A., Mycroft, M., List, M., and Fernández-Juricic, E. (2014). Simulating the responses of forest bird species to multi-use recreational trails. Landscape and Urban Planning, 127, 164–172.

Rohit Kumar, Ankit Pandey, Rekha Rana and Ashutosh Yadav (2019). Climate Change and Mitigation through Agroforestry. Int.J.Curr. Microbiol.App.Sci 8(6): 1662-1667.

Romero-Calcerrada, R. and Luque, S (2006). Habitat quality assessment using Weights-of-Evidence based GIS modelling: The case of Picoides tridactylus as species indicator of the biodiversity value of the Finnish forest. Ecol. Model.196:62–76.

Roointan R, Yadollahi A, Sarikhani Khorami S, Arab MM, Vahdati K (2018) Rainfed fruit orchards in sloping lands: soil erosion reduction, water harvesting and fruit production. Acta Horticulturae 1190: 107-112.

Roots of Peace (2008). Apple production, perennial crop support series Jalalabad, Afghanistan, Publication No. 2008-004-AFG. Available via USAIDAFGHANISTAN.

Rouget M, Richardson DM, Cowling RM, Lloyd JW and Lombard AT. (2003). Current patterns of habitat transformation and future threats to biodiversity in terrestrial ecosystems of the Cape floristic region, South Africa. Biol Conserv. 112:63–85.

Rozenstein, O. and Karnieli, A (2011). Comparison of methods for land use classification incorporating remote sensing and GIS inputs. Appl. Geogr., 31: 533−544.

Saaty T.L (1980). The analytic hierarchy processes. McGraw-Hill: New York.

Saaty, T. L., and Vargas, L. G. (2000). Models, Methods, Concepts and Applications of the Analytic Hierarchy Process. International Series in Operations Research & Management Science. doi:10.1007/978-1-4615-1665-1.

Saaty, T. L. (2003). Decision-making with the AHP: Why is the principal eigenvector necessary? Eur. J. Oper. Res.145 (1)85–9.1.

Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M and Wall, D.H (2000). Biodiversity: global biodiversity scenarios for the year 2100. Science 287:1770–1774.

Sandifer, P.A.; Sutton-Grier, A.E.; Ward, B.P (**2015**). Exploring connections among nature, biodiversity, ecosystem services, and human health and well-being: Opportunities to enhance health and biodiversity conservation. Ecosyst. Serv. 12, 1–15.

Sangam S.; Htut, A.Y (2016). Land use and climate change impacts on the hydrology of the Bago River Basin, Myanmar. Environ. Model. Assess., 21, 819–833.

Schägner, J. P., Brander, L., Paracchini, M. L., Maes, J., Gollnow, F., and Bertzky, B. (2018). Spatial dimensions of recreational ecosystem service values: A review of meta-analyses and a combination of meta-analytic value-transfer and GIS. Ecosystem Services, 31, 395–409.

Schmidt, K., Martín-López, B., Phillips, P. M., Julius, E., Makan, N., and Walz, A. (2019). Key landscape features in the provision of ecosystem services: Insights for management. *Land Use Policy, 82, 353–366.*

Schröter, M., van der Zanden, E. H., van Oudenhoven, A. P. E., Remme, R. P., Serna-Chavez, H. M., de Groot, R. S., & Opdam, P. (2014). *Ecosystem Services as a Contested Concept: a Synthesis of Critique and Counter-Arguments. Conservation Letters, 7(6), 514–523.*

Schwoerer T, and Dawson NG (2022). Small sight—Big might: Economic impact of bird tourism shows opportunities for rural communities and biodiversity conservation. PLoS ONE 17(7): e0268594.

Selemawi Abrehe, Atkilt Girma & Amare Gebremedhin Nigusse (2021). Potential ecotourism site suitability evaluation for sustainable natural resource management in Kafta Sheraro National Park (KSNP), north-western Tigray, Ethiopia, Journal of Ecotourism, 20:4, 341-370, DOI: 10.1080/14724049.2021.1897599

Selamyihun Kidanu (2004). Using Eucalyptus for soil and water conservation on the highland Vertisols of Ethiopia. Ph.D. Thesis. Wageningen University, The Netherlands.

Serra, P., Pons, X., and Saurí, D. (2008). Land-cover and land-use change in a Mediterranean landscape: A spatial analysis of driving forces integrating bio-physical and human factors. Applied Geography, 28 (3), 189–209.

Shahid, M., Rahman, K.U., Haider, S., Hamza Farooq-Gabriel., Abdul Jabbar Khan., Quoc Bao Pham., Chaitanya B. Pande., Nguyen Thi Thuy Linh and Duong Tran Anh (2021). Quantitative assessment of regional land use and climate change impact on runoff across Gilgit watershed. *Environ Earth Sci* **80**, 743.

Sharma N, Sharma S, Verma S, Sharma C (2013). Impact of changing climate on apple production in Kotkhai area of Shimla district, Himachal Pradesh. Int J Farm Sci 3(1):80–89.

Sharma, P.; Tiwari, P.; Verma, K.; Singh, M (2017). Agroforestry Systems: Opportunities and Challenges in India. J. Pharmacogn. Phtochem. 953–957.

Sharp, R., Douglass, J., Wolny, S., Arkema, K., Bernhardt, J., Bierbower, W., Chaumont, N., Denu, D., Fisher, D., Glowinski, K., Griffin, R., Guannel, G., Guerry, A., Johnson, J., Hamel, P., Kennedy, C., Kim, C.K., Lacayo, M., Lonsdorf, E., Mandle, L., Rogers, L., Silver, J., Toft, J., Verutes, G., Vogl, A. L., Wood, S, and Wyatt, K. (2020). InVEST User's Guide. The Natural Capital Project, Stanford University, University of Minnesota, The Nature Conservancy and World Wildlife Fund.

Sherrouse, B.C., Clement, J.M., Semmens, D.J., (2011). A GIS application for assessing, mapping, and quantifying the social values of ecosystem services. Appl. Geogr. 31, 748–760.

Shilong Piao, Philippe Ciais, Yao Huang, ZehaoShen, ShushiPeng, Junsheng Li, Liping Zhou, Hongyan Liu, Yuecun Ma, Yihui Ding, Pierre Friedlingstein, Chunzhen Liu, Kun Tan, Yongqiang Yu, Tianyi Zhang and Jingyun Fang (**2010**). The impacts of climate change on water resources and agriculture in China. Nature 467, 43–51.

Silberman, J.A., and Rees, P.W (2010). Reinventing mountain settlements: A GIS model for identifying possible ski towns in the U.S. Rocky Mountains. Applied Geography, 30(1), 36-49.

Simelton E, Dam VB, Catacutan D, Do TH, Nguyen TH, Traldi R. (2013). Local capacity for implementing payments for environmental services schemes: lessons from the RUPES project in northeastern Viet Nam. Working Paper 163. Hanoi, Vietnam. World Agroforestry Centre (ICRAF), Southeast Asia Regional Program. 34p

Simeneh Admasu (2020). Ecotourism as a Tool to Substantially Funding Conservation Endeavors in Ethiopia. J Tourism Hospit 9:432.

Simeneh Admasu, Hayal Desta, Kumelachew Yeshitela, and Mekuria Argaw (2022). Analysis of Land Suitability for apple-based agroforestry farming in the Dire and Legedadi watersheds of Ethiopia. Implications for ecosystem services. Heliyon; 8: e11217.

Simeneh Admasu, Kumelachew Yeshitela, and Mekuria Argaw (2023a). Impact of Land Use and Land Cover changes on ecosystem service values in the Dire and Legedadi watersheds, central highland of Ethiopia. Heliyon 9: e15352.

Simeneh Admasu, Kumelachew Yeshitela, and Mekuria Argaw (2023b). Assessing habitat quality using the InVEST model in the Dire and Legedadi watersheds, central highland of Ethiopia. Implication for watershed management. Sustainable Environment 9:11.

Simeneh Admasu, Kumelachew Yeshitela, and Mekuria Argaw (2023c). Impacts of Land Use Land Cover Changes and Climate Variabilities on Annual Water Yield in the Dire and Legedadi Watersheds. Water Conservation Science and Engineering. 8:14.

Simeneh Admasu, Kumelachew Yeshitela, and Mekuria Argaw (2023e). Assessing recreational potential in the Dire and Legedadi landscapes. Implications for ecotourism development. (under review).

Sintayehu Hailu Alemu, Luuk van Kempen, Ruerd Ruben (2017). Explaining technical inefficiency and the variation in income from apple adoption in highland Ethiopia: The role of unequal endowments and knowledge asymmetries. Journal of Agriculture and Rural Development in the Tropics and Subtropics; 118:1.

Sisay Habtegebreal Mogesie (2014). Impact of Land use and landcover Change on Soil Erosion Potential in Legedadi Reservoir Watershed. MSc thesis, Addis Ababa University, Addis Ababa.

Sisay Nune Hailemariam, Teshome Soromessa and Demel Teketay (2016). Land Use and Land Cover Change in the Bale Mountain Eco-Region of Ethiopia during 1985 to 2015. Land 5: 41.

Solomon Belay, Aklilu Amsalu and Eyualem Abebe (2014). Land Use and Land Cover Changes in Awash National Park, Ethiopia: Impact of Decentralization on the Use and Management of Resources. Open Journal of Ecology, 4, 950-960.

Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M ,Miller HL (2007). The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the IntergovernmentalPanel on Climate Change. Cambridge, UK and New York, NY, USA: CambridgeUniversity Press; (IPCC Fourth Assessment Report (AR4)).

Solomon Tiruneh, Amare Bantidir, Venusia Gete, Gete Zeleke and Giuseppe Cirella (2019). Willingness to Pay for Watershed Management. Resources, 8; 77.

Song, W.; Deng, X.; Yuan, Y.; Wang, Z.; Li, Z (2015). Impacts of land use change on valued ecosystem service in rapidly urbanized North China Plain. Ecol. Model. 318, 245–253.

Springgay, Ramirez, Janzen and Brito (2019). The Forest-Water Nexus: An International Perspective. Forests. (10)10: 915.

Srivastava, A.; Kumari, N.; Maza, M (2020). Hydrological response to agricultural land use heterogeneity using variable infiltration capacity model. Water Resour. Manag., 34, 3779–3794.

Stave, K., Goshu, G., and Aynalem, S., (2017). Social and ecological system dynamics: characteristics, trends, and integration in the Lake Tana Basin, Ethiopia. Springer, Cham, Switzerland.

Stephen Polasky, Erik Jeremy Nelson, Derric N Pennington and Kris A. Johnson (2011). The impact of land-use changes on ecosystem services, biodiversity and returns to landowners: a case study in the state of Minnesota. Environ. and Resour. Econ., 48(2), 219–242.

Sterner, T. 2003. Policy instruments for environmental and natural resource management. Resources for the Future. Washington DC, USA.

Stolton S, Mansourian S and Dundley N. (2010). Valuing protected areas. World Bank`s Global Environment Facility Coordination Team, World Bank GEF Operations. Washington DC, USA.

Su, X., Liu, Y., Zhou, W., and Liu, G. (2020). An overview on status and importance of Tibetan Plateau steppe, China in Reference Module in Earth Systems and Environmental Sciences, (Amsterdam: Elsevier).

Summers, J.K., Smith, L.M., Case, J.L., and Linthurst, R.A., (2012). A Review of the Elements of Human Well-Being with an Emphasis on the Contribution of Ecosystem Services. Ambio 41, 327-340.

Sun, X.; Jiang, Z.; Liu, F.; Zhang, D (2019). Monitoring spatio-temporal dynamics of habitat quality in Nansihu Lake basin, eastern China, from 1980 to 2015. Ecol. Indic.102, 716–723.

Surafel Alebel and Amare Tura (2022). Evaluation of climate change impact on Legedadi and Dire Reservoir using R-R-V criteria. Research Square; 2022.

Snyder, S.A., Whitmore, J.H., Schneider, I.E., and Becker, D.R. (2008). Ecological criteria, participant preferences, and location models: A GIS approach toward ATV trail planning. Applied Geography, 28(4); 248-258.

Swetnam, R.D., Fisher, B., Mbilinyi, B.P., Munishi, P.K., Willcock, S.,Ricketts, T., Mwakalila, S., Balmford, A., Burgess, N.D., Marshall,A.R. and Lewis, S.L. (2011). Mapping socio-economic scenarios of landcover change: a GIS method to enable ecosystem service modelling. Journal of Environmental Management,92, 563–574.

Taddese H (2014). Suitability analysis for Jatropha curcas production in Ethiopia - a spatial modeling approach. Environ Syst Res 3:25.

Takano, T., Nakamura, K. and Watanabe, M (2002). Urban residential environments and senior citizens' longevity in megacity areas: The importance of walkable green spaces. J. Epidemiol. Community Health, 56, 913–918.

Tallis, H., Mooney, H., Andelman, S., Balvanera, P., Cramer, W., Karp, D., … Walz, A. (2012). A Global System for Monitoring Ecosystem Service Change. *BioScience, 62(11), 977–986.*

Tamirat, G., and Muluken, P., (2018). Analysis of apple fruit value chain in Southern Ethiopia; the Case of Chencha District. J. Plant Breed. Crop Sci. 6, 26–34.

Tamiru Alemayehu., Daganchew Legesse., Tenalem Ayenew., Yirga Tadess., Solomon Waltenigus and Nuri Mohammed (2005). Hydrogeology, water quality and the degree of groundwater vulnerability to pollution in Addis Ababa, Ethiopia. Addis Ababa University press, UNEP/UNESCO report 134pp.

Tang Feng, Meichen Fu, Li Wang and Pengtao Zhang (2020). Land-use change in Changli County, China: Predicting its spatio-temporal evolution in habitat quality. Ecological Indicators,117:106719.

Tatek Belay, Tadele Melese, and Abebe Senamaw (2022). Impacts of land use and land cover change on ecosystem service values in the Afroalpine area of Guna Mountain, Northwest Ethiopia. Heliyon 8 (2022) e1224.

Taye Aduna (2009). Impact of LULC Change on Catchment Hydrology and Water Quality of Legedadi-Dire Catchments. MSc Thesis Addis Ababa University, Addis Ababa.

Tekle W. (2008). Personal communication with the Deputy General Manager of Addis Ababa Water Supply and Sewerage Authority. July 2nd 2008. Addis Ababa.

Temesgen Gashaw, Taffa Tulu, Mekuria Argaw, Abeyou W. Worqlul, Terefe Tolessa, Mengistie Kindu (2018). Estimating the impact of land use/cover change on Ecosystem Service Value in the Andassa watershed in the Upper Blue Nile basin of Ethiopia, Ecosystem Services; 31: 219-228.

Terefe Tolessa, Feyera Senbeta and Tarik Abebe (2016): Land use/ cover analysis and ecosystem services valuation in the central highlands of Ethiopia, Forests, Trees and Livelihoods, 26(2), 111–123.

Terefe Tolessa, Feyera Senbeta and Moge Kidane (2017). The impact of land use/cover change on ecosystem services in the central highlands of Ethiopia. Ecosystem Services 23: 47–54.

Terrado, M.; Sabater, S.; Chaplin-Kramer, B.; Mandle, L.; Ziv, G.; Acuna, V (2016). Model development for the assessment of terrestrial and aquatic habitat quality in conservation planning. Sci. Total Environ. 540, 63–70.

Tesfaye, A., Negatu, W., Brouwer, R., Van der Zaag P. (2014). Understanding soil conservation decision of farmers in the Gedeb watershed, Ethiopia. Land Degradation & Development, 25, 71-79.

Tesfaye, A. and Brouwer, R. (2016). Exploring the scope for transboundary collaboration in the Blue Nile River basin: downstream willingness to pay for upstream land use changes to improve irrigation water supply. *Environment and Development Economics*, 21, pp 180-204.

Tesfaye, A., Brouwer, R., van der Zaag, P., and Negatu, W. (2016). Assessing the costs and benefits of improved land management practices in three watershed areas in Ethiopia. *International Soil and Water Conservation Research, 4(1), 20–29.*

Tesfaye G Admasu and Amene A Jenberu (2022). The Impacts of Apple-based Agroforestry Practices on the Livelihoods of Smallholder Farmers in Southern Ethiopia. Trees, Forests and People 7 (2022) 100205.

The Economics of Ecosystem and Biodiversity (TEEB) (2018). TEEB for Agriculture and Food: Scientific and Economic Foundations. UN Environment, Geneva, Switzerland.

Thomas, H., and Nisbet, T. R. (2007). An assessment of the impact of floodplain woodland on flood flows. *Water and Environment Journal, 21(2): 114–126.*

Thomas E, Jansen M, Chiriboga-Arroyo F, Wadt LHO, Corvera-Gomringer R, Atkinson RJ, Bonser SP, Velasquez-Ramirez MG and Ladd B (2021). Habitat Quality Differentiation and Consequences for Ecosystem Service Provision of an Amazonian Hyperdominant Tree Species. Front. Plant Sci. 12:621064.

Tilman, D., Reich, P. B. and Knops, J. M. H. (2006). Biodiversity and ecosystem stability in a decade-long grassland experiment. Nature 441: 629–632.

Tirusew Asefa and Fekadu Moreda (2023). Addis Ababa's Chronic Urban Water Supply: The Ticking Time Bomb, weAspire.

TNC (2015). Upper Tana-Nairobi Water Fund Business Case. Version 2. The Nature Conservancy: Nairobi, Kenya.

TNC (2016). Sub-Saharan Africa’s Urban Water Blueprint: Securing Water Through Water Funds and Other Investments in Ecological Infrastructure. The Nature Conservancy: Nairobi, Kenya

TNC (2021c). “TNC Water Funds Toolbox.” https://waterfundstoolbox.org/ getting-started/what-is-a-water-fund.

Tolba, M.K. and El-Kholy, O.A (1992). The World Environment 1972–1992: Two Decades of Challenge. Chapman & Hall, London.

Torralba, M., Fagerholm, N., Burgess, P.J., Moreno, G., and Plieninger, T (2016). Do European agroforestry systems enhance biodiversity and ecosystem services? A meta-analysis. Agric. Ecosyst. Environ. 230: 150–161.

Tratalos, J. A., Haines-Young, R., Potschin, M., Fish, R., and Church, A. (2016). Cultural ecosystem services in the UK: Lessons on designing indicators to inform management and policy. *Ecological Indicators, 61, 63–73.*

Troy, A., and Wilson, M. A. (2006). Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer. *Ecological Economics, 60(2), 435–449.*

Tucker, C.J (1979). Red and photographic infrared linear combinations for monitoring vegetation.Remote Sens. Environ. 8, 127–150.

Tyagi, S., Mohan, K. and Kumar, S. (2002). Role of agroforestry in watershed management in the state of Jharkhand. 12th ISCO Conference. Beijing.

Tylianakis, J. M., Didham, R. K., Bascompte, J., and Wardle, D. A. (2008). Global change and species interactions in terrestrial ecosystems. Ecology Letters, 11(12), 1351–1363.

van Berkel, D.B. and Verburg, P.H. (2014). Spatial quantification and valuation of cultural ecosystem services in an agricultural landscape. Ecol. Indic. 37, 163–174.

van der Ploeg S, de Groot RS (2010). The TEEB Valuation Database–a searchable database of 1310 estimates of monetary values of ecosystem services. Wageningen, The Netherlands: Foundation Sustainable Development.

Van Rooijen D. and Taddesse G. (2009). Urban sanitation and wastewater treatment in Addis Ababa in the Awash Basin, Ethiopia. Water, Sanitation and Hygiene: Sustainable Development and Multisectoral Approaches 34th WEDC International Conference, Addis Ababa, Ethiopia.

Vargas, L., Willemen, L., and Hein, L.J.E.M., (2019). Assessing the Capacity of Ecosystems to Supply Ecosystem Services Using Remote Sensing and An Ecosystem Accounting Approach. 63, 1-15.

Verchot L V, Van Noordwijk M, Kandji S, Tomich T, Ong C, Albrecht A, and Palm, C. (2007). Climate change: linking adaptation and mitigation through agroforestry. Mitigation and adaptation strategies for global change, 12(5): 901-918.

Verhagen, W., Van Teeffelen, A. J. A., Baggio Compagnucci, A., Poggio, L., Gimona, A., & Verburg, P. H. (2016). Effects of landscape configuration on mapping ecosystem service capacity: a review of evidence and a case study in Scotland. *Landscape Ecology, 31(7), 1457–1479.*

Villamagna, A. M., Angermeier, P. L., and Bennett, E. M. (2013). Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. Ecological Complexity, 15, 114–121.

Vitousek, P.M., Mooney, H.A., Lubchenco, J and Melillo, J.M (1997). Human domination of earth's ecosystems. Science 277: 494–499.

Vukomanovic, J., and Orr, B. (2014). Landscape Aesthetics and the Scenic Drivers of Amenity Migration in the New West: Naturalness, Visual Scale, and Complexity. Land, 3(2), 390–413.

Wang, G., Mang, S., Cai, H., Liu, S., Zhang, Z., Wajng, L., Innes, J (2016). Integrated watershed management: evolution, development, and emerging trends. *J. For. Res.* **27**, 967–994.

Wendland KJ, Honzák M, Portela R, Vitale B, Rubinoff S, Randrianarisoa J (2010). Targeting and implementing payments for ecosystem services: opportunities for bundling biodiversity conservation with carbon and water services in Madagascar. Ecol Econ. 69(11):2093–107.

Wegner, G. I. (2015). Payments for ecosystem services (PES): a flexible, participatory, and integrated approach for improved conservation and equity outcomes. *Environment, Development and Sustainability, 18(3), 617–644.*

Westholm, L.and Ostwald, M (2020). Food Production and Gender Relations in Multifunctional Landscapes: A Literature Review. Agroforest Syst. 94, 359–374.

Willemen, L., Burkhard, F., Crossman, N., Drakou, E. G., and Palomo, I. (2015). Best practices for mapping ecosystem services: the quest for a green bullet*: PowerPoint*. 1-12.

Willemen, L., Crossman, N.D., Quatrini, S., Egoh, B., Kalaba, F.K., Mbilinyi, B., and de Groot, R., (2017). Identifying ecosystem service hotspots for targeting land degradation neutrality investments in south-eastern Africa. Journal of Arid Environments, 1-12.

Williams WT, and Tax Tzoc BA (1990). Preliminary observations on the effects of hydrogen cyanamide on breaking dormancy and harvest of apples in Guatemala. Acta-Horticulturae 279: 399-408.

Wolff, S., Schulp, C. J. E., & Verburg, P. H. (2015). Mapping ecosystem services demand: A review of current research and future perspectives. *Ecological Indicators, 55, 159–171.*

Wollenberg, E., Tennigkeit, T., Dinesh, D., Baumert, S., Röhrig, F. Kirfel-Rühle, L., Zeppenfeldt, L. (2022). Compensating farmers for ecosystem services: Lessons and an agenda for innovation. CompensACTION Policy Brief.

Wondimagegn Mengist, Teshome Soromessa and Gudina Legese Feyisa (2022). Estimating the total ecosystem services value of Eastern Afromontane Biodiversity Hotspots in response to landscape dynamics. Environmental and Sustainability Indicators 14:100178.

World Bank (2007). The cost of land degradation in Ethiopia: a review of past studies. Washington, DC: World Bank.

World Bank (2007). Project appraisal document on a proposed credit to Ethiopia for an Urban Water Supply and Sanitation Project. Washington.

World Commission on Dams (2000). Dams and development: A new framework for decision-making. The Report of the World Commission on Dams.Earthscan Publications LTD, London.

Wu, W. G., J. K. Huang, and X. Z. Deng (2009). "Potential land for plantation of Jatropha curcas as feedstock for biodiesel in China," *Science in China - Series D: Earth Sciences*, vol. 53, no. 1, pp. 120–127.

Wu, C.-F.; Lin, Y.P.; Chiang, L.C.; Huang, T (2014). Assessing highway's impacts on landscape patterns and ecosystem services: A case study in Puli Township, Taiwan. Landsc. Urban Plan, 128: 60–71.

Wunder, Sven (2006). Are Direct Payments for Environmental Services Spelling Doom for Sustainable Forest Management in the Tropics? Ecology and Society 11(2):23.

Wunder, S., Engel, S. and Pagiola, S. (2008). Taking stock: a comparative analysis of payments for environmental services programs in developed and developing countries. Ecological Economics 65: 843–852.

Yan, S., Wang, X., Cai, Y., Li, C., and Yan, R., (2018). An integrated investigation of spatiotemporal habitat quality dynamics and driving forces in the upper basin of miyun reservoir, north China. Sustainability 10 (4625), 1–17.

Yanan Li, Linghua Duo, Ming Zhang, Jingyuan Yang, Xiaofei Guo (2022). Habitat quality assessment of mining cities based on InVEST model-a case study of Yanshan County, Jiangxi Province International Journal of Coal Science & Technology (2022) 9:28

Yang, D.; Liu, W.; Tang, L.; Chen, L.; Li, X.; Xu, X (2019). Estimation of water provision service for monsoon catchments of South China: Applicability of the InVEST model. Landsc.Urban Plan. 182, 133–143.

Yang Zou and DeHua Mao (2021). Analysis of water yield service of Lianshui River Basin in China based on ecosystem services flow model Water Supply Vol 22 No 1, 335 doi: 10.2166/ws.2021.265.

Yasin, G.; Nawaz, M.F.; Martin, T.A.; Niazi, N.K.; Gul, S.; Yousaf, M.T.B (2019). Evaluation of Agroforestry Carbon Storage Status and Potential in Irrigated Plains of Pakistan. Forests, 10, 640.

Yilikal Anteneh, Gete Zeleke, and Ephrem Gebremariam. (2018). Assessment of surface water quality in Legedadie and Dire catchments, Central Ethiopia, using multivariate statistical analysis. *Acta Ecologica Sinica, 38(2), 81–95.*

Yilikal Anteneh, Gete Zeleke and Ephrem Gebremariam (2019). Valuing the water supply: ecosystem-based potable water supply management for the Legedadie-Dire catchments, Central Ethiopia. Ecological Processes, 8:9.

Ying X, Guang-Ming Z, Gui-Qiu C, Lin T, Ke-Lin W, and Dao-You H (2007). Combining AHP with GIS in synthetic evaluation of eco-environment quality-a case study of Hunan Province, China. Ecol Model 209(2–4):97–109.

Yohannes Ayanu, Conrad, Christopher, Nauss, Thomas, Wegmann, Martin and Koellner, Thomas (2012). Quantifying and Mapping Ecosystem Services Supplies and Demands: A Review of Remote Sensing Applications. Environmental science & technology 46(16):8529-41.

Young, C. E. F., and de Bakker, L. B. (2014). *Payments for ecosystem services from watershed protection: A methodological assessment of the Oasis Project in Brazil. Natureza & Conservação, 12(1), 71–78.*

Yu, J.; Yuan, Y.; Nie, Y.; Ma, E.; Li, H.; Geng, X (2015). The temporal and spatial evolution of water yield in Dali County.Sustainability 7, 6069–6085.

Yuan, F., Sawaya, K. E., Loeffelholz, B. C., and Bauer, M. E. (2005). Land cover classification and change analysis of the Twin Cities (Minnesota) Metropolitan Area by multitemporal Landsat remote sensing. Remote Sensing of Environment, 98(2–3): 317–328.

Yue Li., ShilongPiao., Laurent Z. X. Li., Anping Chen., Xuhui Wang., Philippe Ciais., Ling Huang., XuLian., ShushiPeng., ZhenzhongZeng., Kai Wang., and Liming Zhou (2018). Divergent hydrological response to large-scale afforestation and vegetation greening in China. Sci. Adv., 4, eaar4182.

Zahoor, S., Dutt, V., Mughal, A. H., Pala, N. A., Qaisar, K. N., and Khan, P. A. (2021). Apple-based agroforestry systems for biomass production and carbon sequestration: implication for food security and climate change contemplates in temperate region of Northern Himalaya, India. Agroforestry Systems, 95(2), 367–382.

Zardari, N. H, K. Ahmed, S. M. Shirazi, and Z. Yusop (2015). Weighting Methods and their Effect on Multi-Criterial Decision-Making Model Outcomes in Water Resources Management, Springer, Berlin, Germany.

Zasada, I (2011). Multifunctional peri-urban agriculture-A review of societal demands and the provision of goods and services by farming. Land Use Policy 28, 639-648.

Zavala G. C. (2004). Temperature effects on fruit and shoot growth in apple (Malus domestica) early in the season," *Acta Horticulture*, vol. 636, no. 636, pp. 447–453.

Zerihun Nigussie, Getachew Fisseha, Getachew Alemayehu, Steffen Abele (2019). Smallholders' apple-based agroforestry systems in the north-western highlands of Ethiopia. Agroforest Syst 93:1045–1056.

Zhang, L., Dawes, W.R., Walker, G.R. (2001). Response of mean annual evapotranspiration to vegetation changes at catchment scale. Water Res. Res. 37, 701–708.

Zhang, Y.; Qi, W.; Zhou, C.; Ding, M.; Liu, L.; Gao, J.; Bai, W.; Wang, Z.; Zheng, D (2014). Spatial and temporal variability in the net primary production of alpine grassland on the Tibetan Plateau since 1982. J. Geogr. Sci. 24, 269–287.

Zhang, L.; Cheng, L.; Chiew, F.; Fu, B (2018). Understanding the impacts of climate and landuse change on water yield. Curr.Opin. Environ. Sustain. 33, 167–174.

Zhang, H., Zhang, C., Hu, T., Zhang, M., Ren, X., and Hou, L. (2020). Exploration of roadway factor and habitat quality in InVEST. Trans. Res. D Trans. Environ. 87:102551.

Zhu, C., Zhang, X., Zhou, M., He, S., Gan, M., Yang, L., and Wang, K. (2020). Impacts of urbanization and landscape pattern on habitat quality using OLS and GWR models in Hangzhou, China. *Ecological Indicators, 117: 106654.*

Zhang, Y.; Zhang, C.; Zhang, X.; Wang, X.; Liu, T.; Li, Z.; Lin, Q.; Jing, Z.; Wang, X.; Huang, Q.; Wenxin Sun; Jun Zhai; Li Tan; Jieqing Wang; Guoyan Zhou; Yasi Tian; Jianli Hao; Yu Song, and Fei Ma (2022). Habitat quality Assessment and Ecological Risks Prediction: An Analysis in the Beijing-Hangzhou Grand Canal (Suzhou Section). Water 14, 2602.

Zhou, W.; Liu, G.; Pan, J.; Feng, X (2005). Distribution of available soil water capacity in China. J. Geogr. Sci. 15, 3–12.

Zhu, L.; Huang, C.; Liu, Q.S.; Liu, G.H (2015). Changes of coastal zone landscape spatial patterns and ecological quality in Liaoning province from 2000 to 2010. Resour. Sci. 2015, 37, 1962–1972.

Zinabu Assefa and Michael O. Dioha (2020). Modelling scenarios for sustainable water supply and demand in Addis Ababa city, Ethiopia. Environ Syst Res (2020) 9:7.

Zomer, R.J.; Neufeldt, H.; Xu, J.; Ahrends, A.; Bossio, D.; Trabucco, A.; van Noordwijk, M.; Wang, M (2016). Global Tree Cover and Biomass Carbon on Agricultural Land: The Contribution of Agroforestry to Global and National Carbon Budgets. Sci. Rep. 6, 29987.

Zotarelli, L., Dukes, M.D., Romero, C.C., Migliaccio, K.W., Kelly, T., (2015). Step by Step Calculation of the Penman-Monteith Evapotranspiration (FAO-56 Method). 1. UF/IFAS Extension Service, University of Florida, USA, pp. 1–10.

Annex 1: Data collection questionnaire for Habitat quality assessment in the Dire and Legedadi watersheds), central highlands of Ethiopia

I am called Simeneh Admasu conducting a doctoral study in Environmental Planning at the Ethiopian Institute of Architecture, Building Construction and City Development (*EiABC*) of

Addis Ababa University. My doctoral research revolves around quantifying and mapping the impact of LULC changes in the ecosystem services in the Big Upper Akaki catchment (Dire and Legedadi watersheds). This questionnaire is divided into two sections and will take about 15-20 minutes to complete.

Warm regards,

Simeneh Admasu

Section I. Expert and/or KII

A. General Information

a) Name__________ ____________________Age ______Sex ____Education--------------

b) Village__________________________ ____________________________

c) Major occupation________________________________

d) How long have you lived in the area? ________________________________

1. What are the challenges for land management in your area?

----

2. What are the habitat types in your locality?

----

3. What are the most important threats that affect sustainable management of various land uses?

a) --------------------------------

b) ---------------------------------

c) --------------------------------

d) --------------------------------

e) ----------------------------------

f) ---------------------------------

4. How are these threats affecting the biodiversity of the area?

a) --------------------------------

b) ---------------------------------

c) ---------------------------------

d) --------------------------------

e) ----------------------------------

f) ---------------------------------

5. What practices are applied for reducing threats on biodiversity and ecosystems and to improve land productivity?

----

6. How far the threats are situated from the habitat/land uses? (Maximum distance)

----

7. Which threat is distributed quickly to the surrounding habitat/ecosystem? (Exponential)

----

8. Which threat is slowly disturbed that brings huge long-term effect on the ecosystem? (Linear)

----

9. What do you think the contribution of pristine habit for ecosystem services?

10. All the necessary information will be filled by the expert.

No.	Habitat types/land uses	Threat	Maximum distance b/n habitat and the threat	Distance decay*	Remarks

**Distance decay = does the threat has long and short-term effect?*

11. Sensitivity analysis of the threats**.** Based on the Likert scale, a threat may be impact to the habitat quality as 5 (very high), 4 (pose a high threat), 3 (pose a medium threat), 2 (pose a low threat), 1 (poses no threat) and 0 (do not know).

Table 1. Expert judgment matrix format. This *value will be provided by expert judgment based on professional experience by comparison one form the other (comparing two threat impact)*

Threats	Scale					
	5 (very high)	4 (high)	3 (medium)	2 (low)	1 (threat)	0 (do not know)