TECHNISCHE UNIVERSITÄT MÜNCHEN

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Landscape Level Modelling of the Ethiopian Highland Resources -

A geo-informatics application to their sustainable management, use and conservation

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Abstract

Monitoring of land use/land cover (LULC) changes provides critical inputs to evaluate complex causes and responses in order to project future trends better, and it is a prerequisite for making effective development plans. This thesis aims to develop a new methodological framework using geo-informatics for sustainable natural resource management, use and conservation in the Ethiopian highlands from a novel multidisciplinary perspective by taking Munessa-Shashemene landscape as a case study site. Satellite images of Landsat MSS (1973), TM (1986), ETM+ (2000) and RapidEye (2012) were used to derive nine LULC types using object-based image classification. Other datasets required for the study were generated from both primary and secondary sources. Combination of techniques, including post classification comparison, GIS-based processing, descriptive statistics and logistic regression were employed for data analyses of LULC changes of the past four decades (1973-2012) and their drivers. Estimation and change analyses of ecosystem service values (ESVs) were conducted, mainly, by employing GIS using LULC datasets of each reference year with their corresponding global value coefficients developed earlier and own modified conservative value coefficients for the studied landscape. Possible future LULC patterns and changes covering the next four decades (2012-2050) were simulated and examined by using a spatially explicit GIS-based model. Three alternative scenarios, namely Business As Usual (BAU), Forest Conservation and Water Protection (FCWP) and Sustainable Intensification (SI) were used. The classification result revealed that grasslands (42.3%), natural forests (21%), and woodlands (11.4%) were dominant LULC types in 1973. In 2012, croplands (48.5%) were the major LULC types followed by others. The change results showed that about 60% of the land had experienced changes in LULC over the past four decades. Specifically, about 95% of woodlands, 74% of grasslands and 59% of natural forests that existed in 1973 have been converted to other LULCs types. On the other hand, croplands showed rapid expansion of about 272% during the study years. The LULC changes were triggered by the interplay between more than twelve drivers related to social, economic, environmental, policy/institutional and technological factors. Six of them were the top important drivers as viewed by the local people and confirmed by quantitative analyses. As a result of the changes, the study revealed a total loss of ESVs ranging from US$ 19.3 million per year when using own modified value coefficients to US$ 45.9 million per year when employing global value coefficients. The simulation results also showed that areas of croplands will increase widely under the BAU scenario and would expand to the remaining woodlands, natural forests and grasslands, reflecting vulnerability of these LULC types and potential loss of associated ESVs. FCWP scenario would bring competition among other LULC types, particularly more pressure on the grassland ecosystem. The SI scenario, with holistic landscape management approach, demonstrated that expansion of croplands could vigorously be reduced, remaining forests would be better conserved and degraded land would be recovered, resulting in gains of the associated total ESVs. The approach framed in this study is an important tool for supporting appropriate management options of natural resources at the landscape level.

Zusammenfassung

Die Beobachtung von Veränderungen der Landnutzung/Landbedeckung (LN/LB) hilft, die komplexen Zusammenhänge aus Ursache und Wirkung zu verstehen. Dies ist wichtig, um die zukünftigen Entwicklungen besser prognostizieren zu können. Die Beobachtungen sind eine notwendige Voraussetzung für eine erfolgreiche Entwicklungsplanung. Ziel der vorliegenden Arbeit war es, Geoinformatik in einem neuartigen multidisziplinären Ansatz zu verwenden, um damit ein neues methodisches Rahmenwerk für eine nachhaltige Bewirtschaftung der natürlichen Ressourcen – deren Nutzung sowie Schutz – im äthiopischen Hochland zu erstellen. Als Untersuchungsgebiet diente die Munessa-Shashemene-Region. Als Datengrundlage dienten unter anderem Satellitenaufnahmen von Landsat MSS (1973), TM (1986), ETM+ (2000) und RapidEye (2012), wobei unter Verwendung des objekt-basierten Bildklassifikationsansatzes neun verschiedene LULC-Klassen ausgewiesen wurden. Weitere Primär- und Sekundärquellen dienten als Grundlage für zusätzlich benötigte Datensätze. Für die Erfassung und Untersuchung der LN/LB-Veränderungen in den letzten vier Jahrzehnten (1973-2012) sowie deren Ursachen kam eine Kombination unterschiedlicher Methoden zum Einsatz – unter anderem post-classification-comparison, GIS-basierte Analysen, deskriptive Statistik und logistische Regression. Für die Bewertung von Ökosystemdienstleistungen sowie die Beurteilung der Veränderungen wurden mittels GIS-Analysen sogenannte Ecosystem Service Values (ESVs) ermittelt. Als Eingangsdaten wurden hier die LN/LB-Datensätze der jeweiligen Bezugsjahre in Kombination mit zuvor entwickelten Wertkoeffizienten verwendet – sowohl passende globale Koeffizienten als auch eigens an das Untersuchungsgebiet angepasste konservative Koeffizienten. Mögliche LN/LB -Muster und -Veränderungen über die kommenden vier Jahrzehnten (2012-2050) wurden anhand eines räumlich expliziten, GIS-basierten Modells simuliert. Untersucht wurden drei unterschiedliche Szenarien: „Weiter wie bisher" („Business as usual" (BAU)), „Vorrang für Wald- und Wasserschutz" („Forest Conservation and Water Protection" (FCWP)) und „Nachhaltige Intensivierung" („Sustainable Intensification" (SI)). Das Klassifikationsergebnis zeigt, dass 1973 v.a. Grasland (42,3%), natürlicher Wald (21%) und Akazienwälder (11,4%) die vorherrschenden LN/LB-Klassen waren. Im Jahr 2012 war hingegen Ackerland (48,5%) die dominierende LN/LB-Klasse, gefolgt von anderen. Die Untersuchungsergebnisse zeigten, dass sich die vorherrschende LN/LB-Klasse auf ca. 60% der Untersuchungsfläche in den letzten vier Jahrzehnten verändert hat. Von den 1973 existierenden LN/LB-Klassen waren die Klassen natürlicher Wald, Grasland und Akazienwälder am stärksten von den Veränderungen betroffen, wobei ca. 95%, 74% und 59% der Fläche in andere LN/LB-Klassen umgewandelt wurden. Im Gegensatz dazu erweiterte sich der Anteil an Ackerflächen im Untersuchungszeitraum rapide um etwa 272%. Unterschiedlichste soziale, wirtschaftliche, ökologische, politische und technologische Faktoren sowie deren Interaktionen können als Triebfedern für die beobachteten LN/LB-Veränderungen ausgemacht werden. Die einheimische Bevölkerung benannte sechs Faktoren als die wichtigsten Ursachen der Veränderung, was durch eine quantitative

Analyse bestätigt werden konnte. Die LN/LB-Veränderungen während des Untersuchungszeitraums bedingten einen enormen Verlust der vorhandenen Ökosystemdienstleistungen, der unter Verwendung der globalen bzw. der modifizierten konservativen Wertkoeffizienten auf 45,9 bzw. 19,3 Millionen US-Dollar pro Jahr beziffert werden kann. Die Simulation mit unterschiedlichen Szenarios zeigte Folgendes: Im BAU-Szenario nimmt die Ackerfläche weiter stark zu und verdrängt verbleibende Wald- und Urwaldflächen sowie Grasland. Das verdeutlicht die Vulnerabilität dieser LN/LB-Klassen und den damit verbundenen potentiellen Verlust an ESV. Unter einem FCWP-Szenario würde der Druck auf die Flächen mit nicht explizit geschützten LN/LB-Klassen, insbesondere das Grasland, weiter steigen. Wird das SI-Szenario – mit einem ganzheitlich integrierten Ansatz für das Landschaftsmanagement – für die Simulation herangezogen, kann die weitere Ausdehnung der Ackerflächen deutlich reduziert werden. Verbleibende Waldgebiete können besser geschützt werden und aktuell degradiertes Land kann sich regenerieren, was zu einem Anstieg der damit verbundenen ESVs führt. Der in dieser Arbeit umrissene Ansatz ist ein wichtiges Werkzeug, um geeignete Maßnahmen für das Management natürlicher Ressourcen auf Landschaftsebene zu unterstützen.

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1 Introduction

1.1 Background

Natural resources or ecosystems, particularly land use/land cover (LULC) types are available for sustaining life on earth by providing a large number of goods and services. Therefore, they are important concerns in many regions of the world (Niedertscheider et al., 2014; Belward and Skøien, 2015; Taelman et al., 2016). In the second half of the twentieth century, humans have degraded them more rapidly than any time in humankind history, mainly to meet increasing demands as a result of growing world population (MEA, 2005). Nowadays, feeding many people while keeping the environmental integrity in an increasingly uncertain economic situation is one of the major challenges mentioned in global agenda and remains as an important issue in developing countries (Sleeter et al., 2013; Tigabu et al., 2014). The great challenge for the coming decades will be the task of increasing food production to ensure food security for the steadily growing world population, particularly for societies hosted in environmentally vulnerable areas, such as Africa (WSFS, 2009; UN DESA, 2013). The situation is severe in Ethiopia where the foundation of the national economy is agriculture (Garedew et al., 2012).

Ethiopia is one of the largest countries in the Eastern horn of Africa with a total area of 1.13 million km^2 (EMA, 1988) and the continent's most populous nation after Nigeria. According to the Ethiopian national population and housing census of 2007, the total human population was 74 million (CSA, 2007). With an estimated annual growth of 2.7%, the population is projected to have increased to about 90 million in 2015 (CSA, 2013). The country is characterized by diverse topographic features with high and rugged mountains, flat-topped plateaux and deep gorges, incised river valleys and rolling plains. Altitudes range from the highest peak at Ras Dejen, 4620 m above sea level, down to the depression of the Kobat Sink (Afar Depression), about 110 m below sea level (EMA, 1988). These have made possible the presence of diverse resources of fauna and flora, and placed the country as one of the 12 Vavilov Centers of Crop Genetic Diversity (EPA, 1997). Ethiopian agriculture contributes nearly half of the Gross Domestic Product (GDP), 90% of the export revenue and employs about 84% of the country's population (CSA, 2013).

More than 90% of Ethiopia's population lives in the highlands (above 1500 m), which account for nearly 44% of the country's landmass and are endowed with moderate temperatures, and adequate rainfall (800 – 2200mm per annum) for rain-fed agriculture (Hurni et al., 2005). Population density in the Ethiopian highlands is one of the highest in Africa. In these highlands, there is also about 93% of the regularly cultivated land, around 75% of the country's livestock and over 90% of the country's economic activity. As a result, the highlands have been settled and cultivated for millennia (McCann, 1995; Eshetu and Högberg, 2000). Consequently, these highlands are significantly affected by resource depletion. Steep and very steep slopes areas are under crop cultivation (Zeleke and Hurni, 2001). These cultivated lands have suffered from loss of top soil as a result of soil erosion. The severity of such soil erosion is visible from the thick mass of soil taken away by major rivers, such as Abay (the Blue Nile - the longest river in the world), Awash, Omo and Baro (Adimassu et al., 2013). These rivers are nowadays colored into chocolates or dirty brown during the main rain season due to soil erosion from their catchment areas.

Historically, Ethiopia was food self-sufficient and exporter of grains until the late 1950s (Dercon, 1999). Afterwards, however, domestic food production has failed to meet national food requirements as its growth lingered behind the population growth (Holden and Shiferaw, 2004; Abebaw et al., 2010). This means that per capita food availability has been falling, and the number of food insecure people has been increasing. Thus, Ethiopia is currently one of the most food-insecure countries in the world (Abebaw et al., 2010). The government of Ethiopia has developed a number of strategies aiming at improving agricultural productivity for food self-sufficiency and as a driving force for the transformation of national economic development. As a result, the country has been following an Agricultural Development-Led Industrialization (ADLI) policy since mid-1990s (NCSS, 1993). In recent years, within the framework of ADLI, various strategies and programmes (e.g. the Sustainable Development and Poverty Reduction Programme – also called SDPRP (MoFED, 2004), the Plan for Accelerated and Sustained Development to End Poverty (PASDEP; MoFED, 2006), the Productive Safety Net Programme (PNSP; Gilligan et al., 2009), and the Growth and Transformation Plan (MoFED, 2010)) have been implemented. Whereas encouraging results have been achieved with implementation of these strategies and programmes, there are still fundamental challenges to a sustained realization of the full potential of the Ethiopian natural resources (Lemenih and Kassa, 2011; Coria and Sterner, 2011; Amessie, 2014). Unlike in many other parts of the world, the shift from area expansion towards agricultural intensification has not happened in the country’s landscape (Tigabu et al., 2014).

The use and management of natural resources, and returning the vast degraded landscapes to protective and/or productive systems, have also substantial importance to attain the goal of sustainable development in Ethiopia (Lemenih and Teketay, 2004a; Tigabu et al., 2014). This, in turn, requires an understanding of the dynamics in time and space of these resources. In this regard, the importance of spatial data monitoring and evaluation for proper management of globally and locally important natural resources is critical. In recent decades, the application of geo-informatics (GIS and remote sensing) not only revolutionized the way data has been collected, but also significantly improved the quality and accessibility of important spatial information for natural resources management and conservation (Lillesand and Kiefer, 2000; Singh et al., 2012; Nguyen et al., 2015). This is a big step forward towards monitoring global biodiversity and supporting the efforts of national and regional natural resource use and conservation. Although datasets generated based on geo-informatics technology are becoming basic tools for the day-to-day activities of natural resource managers, ecologists, conservationists and others, their full potential and reliability are still unused in many of resource use and conservation programmes (Carver et al., 2012; Jackson et al., 2013). Given the fact that agriculture remains as the main source of livelihoods for rural communities and the on-going natural resource degradation continues unabated in Ethiopia, a research-backed alternative strategy for natural resource management using geo-informatics technology is crucial to provide sustainable livelihoods to the people as well as promote sustainable management, conservation and utilization of natural resources, hence, ensuring healthy future.

1.2 Research problem

There are considerable scientific challenges in facilitating sustainable development while safeguarding natural resource depletion in the Ethiopian highlands. The big question is always from which point to start with? Among the many obvious natural resources-related problems in these highlands are the wide spread changes in LULC as a result of drivers and their effects on ecosystem services. Although these changes result from the necessity to meet locally defined needs, they have also regional and global impacts (Turner et al., 1997; Sleeter et al., 2013).

The current widespread LULC dynamics prevailing in the highlands of Ethiopia are, however, not given sufficient attention. The major gap includes absence of land use planning strongly backed with research that can contribute for sustainable use and conservation of natural resources. For example, land is allocated for various uses with no detail analyses of subsequent consequences. Despite the general recognition of the problem of LULC changes and its impact on agricultural productivity, only few scientific studies have been conducted in some areas of the highlands to provide precise quantitative information on the extent and trends of changes. These studies, however, have shown heterogeneity in direction, pattern, type, and/or magnitude of LULC changes (e.g. Abate, 1994, Rembold, et al., 2000; Zeleke and Hurni, 2001; Bewket, 2002; Dwivedi, et al., 2005; Shiferaw et al., 2011; Wondie et al., 2011). Hence, findings of LULC changes in one area cannot necessarily be replicated in another area. The challenge was also to identify the drivers of such LULC changes obtained through remote sensing techniques, which mainly generate the extent of LULC changes but do not provide explanations about the underlying reasons responsible for the observed changes (Wondie et al., 2011). Thus, attempts exist focusing on drivers of LULC changes as observed from earlier studies (e.g. Tekle and Hedlund, 2000; Bewket 2002; Tegene, 2002; Dessie and Kleman, 2007; Tefera, 2011). However, those research findings suggest that the drivers of LULC changes vary from place to place depending on location-specific factors. A comprehensive study of drivers would be useful to better understand the process of changes for appropriate intervention strategy.

Besides studying LULC dynamics and their drivers, a thorough understanding of the consequences of these changes on ecosystem service values (ESVs) through quantitative knowledge is limited although they are vital in raising awareness of the consequences and improved decision making. Throughout this thesis the term ‘‘ecosystem service values’’ is used in a broader sense, defined as values of ecosystem services in monitory units (Zhao et al., 2004; Hu et al., 2008; Polasky et al., 2011). Dynamics of LULC may increase the provision of some services while decreasing others that affect the ability of biological systems to support human needs, indicating ecological degradation (Kreuter et al., 2001; Polasky et al., 2011), or may cause vice versa. Though many case studies on ESVs have been carried out (e.g. Hein et al., 2006; Barral et al., 2012; Leh et al., 2013; Wang et al., 2014), few have paid enough attention to long-term changes of ESVs in response to LULC changes. While efforts have been made to estimate ESVs and suggest options for areas with scarce data, such studies also lack for many other countries in Africa that have dramatic LULC dynamics (Leh et al., 2013), such as Ethiopia. Moreover, future LULC patterns and changes are still poorly understood in the Ethiopian landscapes even if they are crucial for development of effective land use planning strategies (Serneels and Lambin, 2001). Detailed and meaningful scientific information regarding their opportunities and constraints is still lacking. In this regard, flexible spatial simulation models with the capacity to develop insights into LULC dynamics as well as explore "what if" scenarios are required. GIS can provide such extensive opportunities to model LULC dynamics in a spatially explicit manner (Nourqolipour et al., 2015; Qiang and Lam, 2015). These can expose unwanted developments in the land use system, enabling anticipation of these developments through land use planning.

Thus, results of various studies have demonstrated the need to not only discover but also to understand the complex interrelations between land use/land cover changes, their drivers, effects on ecosystem services and their possible future patterns, and changes in order to guide improved natural resource management in an integrated way. This study, therefore, employed to fill these gaps in a holistic approach using the applications of geo-informatics, retrospectively and prospectively, to provide scientific bases for better management of natural resources in the Ethiopian highlands by taking Munessa-Shashemene landscape as a case study site. The rationales for selection of the study landscape were multiple. Firstly, it is a typical Ethiopian highland in terms of various environmental attributes, such as soils, water, climate and topography. Secondly, the landscape is characterized by mosaics of LULC types that provide different opportunities, but currently suffering from rapid conversions (Lemenih et al., 2005). Thirdly, studies covering other issues have already been conducted on the study landscape (e.g. Abate, 2004; Lemenih, 2004; Lemenih et al., 2005; Fritzsche et al., 2007; Tesfaye et al., 2010) some of which provided a valuable foundation for this study. Besides, no study has been carried out in this particular landscape so far on the research problems addressed in this study for the proper management of natural resources. Thus, this research is paramount important for the study landscape in particular and the Ethiopian highlands in general. The approaches and results of the study, being conducted in a data scare tropical region, could also contribute to other similar places around the world where pressure on natural resources will remain crucial.

1.3 Research objectives and hypotheses

The main objective of this study was to develop a new methodological framework using geo-informatics for sustainable natural resource management, use and conservation in the Ethiopian highlands from a novel multidisciplinary perspective. This is intended, ultimately, to serve as a 'stepping stone' to provide empirical grounds required to make informed decisions in the development of management strategies in a holistic approach. The specific objectives of the study were defined to:

1. analyze and evaluate extent and trends of land use/cover (LULC) changes;
2. identify the drivers for the prevailing trends of LULC changes;
3. estimate and quantify changes in ecosystem service values in response to the prevalent LULC dynamics; and
4. simulate and examine the possible future LULC patterns and changes using scenario modelling likely to prevail in affecting their sustainable management, use and conservation.

The following sets of working hypotheses were formulated:

- land use/land cover types have changed and followed certain patterns for the past four decades (1973-2012);
- land use/land cover changes are triggered by the interplay of multiple drivers behind the process;
- wide spread changes of land use/land cover types lead to alter ecosystem services of landscapes; and
- when holistic landscape management is considered with appropriate framework, expected agricultural production can rather be insured while safeguarding the environment than when attempting only strict implementation of conservation and protection policies.

1.4 Embedded original publications and author's contribution

This study investigated, in detail, and developed a novel framework for an improved approach for sustainable land use systems in the Ethiopian highlands based on four components: (1) LULC changes, (2) understanding their drivers, (3) estimating and quantifying changes in ecosystem service values, and (4) modelling future LULC changes and patterns (Figure 1.1). Each of the components were dealt thoroughly and structured as individual article with their complex interrelations. The titles and short summaries and contributions of authors of these articles are briefly presented below.

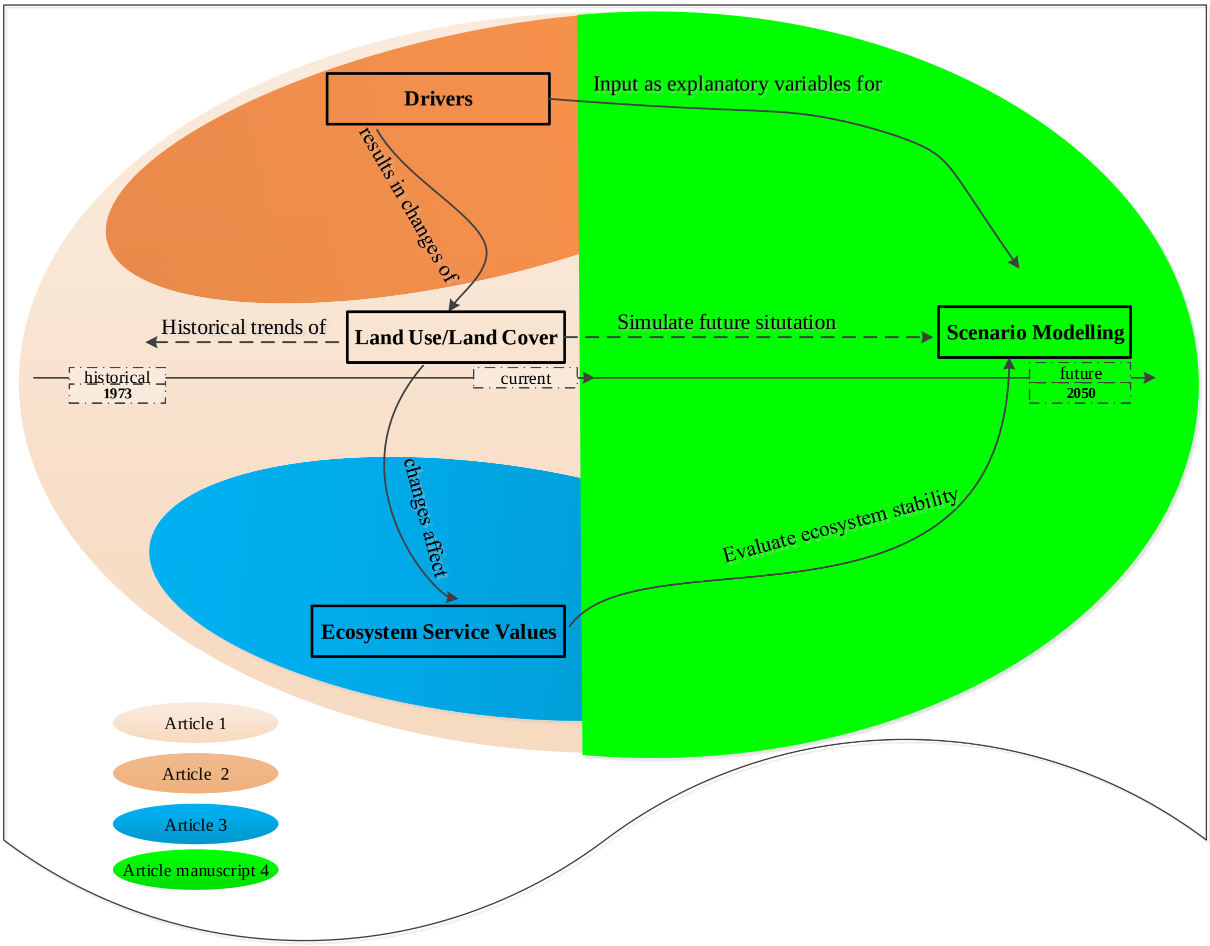

Figure 1.1: Schematic framework of this research with the four main components (solid line boxes) along with their relationship to each other and their coverage in the individual articles.

First article: Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2013: Land Use/Land Cover Change Analysis Using Object-Based Classification Approach in Munessa-Shashemene Landscape of the Ethiopian Highlands. Remote Sensing 5 (5): 2411-2435. doi:10.3390/rs5052411

This work analyzed land use/land cover (LULC) changes in the landscape of Munessa-Shashemene area of the Ethiopian highlands over a period of 39 years (1973–2012). Satellite images of Landsat MSS (1973), TM (1986), ETM+ (2000), and RapidEye (2012) were used.

All images were classified using object-based image classification technique. Change analysis was carried out using post classification comparison in GIS. The result depicted the relationship among LULC types and the changes in different time periods, which served as a base for the other three components of this study. Implication of the findings and the way forward were suggested.

Contributions of authors: The research approach was developed by M. Kindu with the assistance from T. Schneider, D. Teketay and T. Knoke. RapidEye data procurement with the RapidEye Science Archive (RESA) of German Aerospace Center (DLR) project no. 463 was carried out by M. Kindu and T. Schneider. Landsat images were procured by M. Kindu. Field data collection and analyses were carried out by M. Kindu under the supervision of T. Schneider, D. Teketay and T. Knoke. The article was prepared by M. Kindu, and T. Schneider, D. Teketay and T. Knoke have undertaken the article revision and research coordination.

Second article: Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2015: Drivers of land use/land cover changes in Munessa-Shashemene landscape of the south-central highlands of Ethiopia. Environmental Monitoring and Assessment Volume 187, Issue 7, 23 July 2015, Article number 452: 17. doi: 10.1007/s10661-015-4671-7.

This work identified the main drivers behind the LULC changes that had occurred in the past four decades in the same study landscape. The datasets required for the study were generated through both primary and secondary sources. Combination of techniques, including descriptive statistics, GIS-based processing and regression analyses were employed for data analyses. The identified drivers helped to better understand the process of LULC changes and served as inputs for modelling of future changes. The findings can also be useful for making informed decision during policy formulation or land use planning processes or other similar studies in the Ethiopian highlands.

Contributions of authors: The research approach was developed by M. Kindu with the assistance from T. Schneider, D. Teketay and T. Knoke. Field data collection and analyses were carried out by M. Kindu under the supervisions of T. Schneider, D. Teketay and T. Knoke. The article was prepared by M. Kindu, and T. Schneider, D. Teketay and T. Knoke have undertaken the article revision and research coordination.

Third article: Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. (2016): Changes of ecosystem service values in response to land use/land cover dynamics in Munessa–Shashemene landscape of the Ethiopian highlands. Science of the Total Environment 547: 137-147. doi:10.1016/j.scitotenv.2015.12.127

This study estimated changes of ESVs in response to LULC dynamics over the past four decades in the studied landscape, mainly, by employing GIS. The estimated ESV changes were derived from LULC datasets as proxy measurement with their corresponding value coefficients developed earlier by other scientists and own modified conservative value coefficients. It was a new attempt to increase the credibility of estimation for data scarce regions. The study brought valuable empirical evidences that can serve as powerful and arguably essential communication tool to inform higher officials better with regard to trade-offs involved in land resource use options.

Contributions of authors: The research approach was developed by M. Kindu with the assistance from T. Schneider, D. Teketay and T. Knoke. Data collection and analysis was done by M. Kindu under the supervisions of T. Schneider, D. Teketay and T. Knoke. The article was prepared M. Kindu, and T. Schneider, D. Teketay and T. Knoke have undertaken the article revision and research coordination.

Fourth article: Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. (manuscript): Scenario Modelling of Land Use/Land Cover Changes in Munessa-Shashemene Landscape of the Ethiopian Highlands.

In this work, a spatially explicit GIS based model was developed to simulate possible future LULC patterns and changes in three scenarios in the same study landscape covering four decades (2012-2050) for the first time under three ‘‘what if’’ scenarios. Own generated datasets from the three components were the main sources to conduct this particular study. The findings are intended to serve as an early warning system for understanding the future effects of LULC dynamics and serve as a strategic guide to land use planning process that can better balance agricultural production and ecological conservations.

Contributions of authors: The research approach was developed by M. Kindu with the assistance from T. Schneider, D. Teketay and T. Knoke. Data collection and analysis were carried out by M. Kindu under the supervision of T. Schneider, D. Teketay and T. Knoke. The manuscript was prepared by M. Kindu, and T. Schneider, D. Teketay and T. Knoke have undertaken the manuscript revision and research coordination.

1.5 Organization of the thesis

The thesis is organized into six chapters. The first chapter (this chapter) introduces the background, research problem, main objectives with hypotheses, and embedded original publications with contribution of authors. Chapter two reviews the state of the art, i.e. geo-informatics applications for studying LULC dynamics, understanding their drivers, quantifying ecosystem services in relation to the changes, and scenario modelling of future land use/land cover patterns and changes. Chapter three describes the study area and the general methodology employed for data collection and analyses. Chapter four is devoted to present the major findings in sub-sections. Chapter five brings the results together and discusses them systematically. The last chapter (Chapter six) concludes by presenting a summary of major findings and highlighting future area of investigation.

2 State of the art

2.1 Geo-informatics for the study of extent and dynamics of LULC

LULC types and their associated changes were central to all related sustainable development issues (Turner, 1997), and, yet, today they are also important concerns in many regions of the world (Sleeter et al., 2013; Niedertscheider et al., 2014; Belward and Skøien, 2015; Taelman et al., 2016). Their changes have many interactions with other components of the earth system. It is recognized that dramatic changes in LULC can significantly modify regional climate (Fairman et al., 2011) and water balance (Davis et al., 2015), silt-up streams (Zaimes and Schultz, 2015), affect biodiversity (Dayamba et al., 2016) and ecosystem stability (Paz-Kagan et al., 2014), and disrupt socio-cultural practices (Raynaut, 2001). This provides an important aspect in evaluating and conserving Earth's resources and, thus, calls for global attention for continuous monitoring of the changes.

Up-to-date datasets on LULC change provide critical inputs to evaluate complex causes and responses in order to project future trends better, ranging from local to global scales (Prenzel, 2004; Giri, 2005). They are also prerequisites for making development plans (Gautam and Chennaiah, 1985). Various studies have been conducted all over the world regarding LULC changes of a given landscape. However, the magnitude of LULC change differs with the time period being examined (Weng, 2002), geographical location (Rindfuss et al., 2004), slope gradient and elevation range (Nelson and Geoghegan, 2002).

Results of various studies have demonstrated the need for a study focusing on location-specific LULC changes for sound decision-making processes related to the use and conservation of natural resources (Bewket, 2002; Lunetta et al., 2006; Wondie et al., 2011; Xiuqin et al., 2011). Two ways of capturing LULC dynamics are available: conventional ground- and geo-informatics (remote sensing and GIS) based methods. The ground method is labor intensive, time consuming and difficult for capturing data from inaccessible areas with ragged topographies like the case of most Ethiopian landscapes. On the contrary, with the advent and development of the integrated geospatial techniques that integrate the use of remote sensing and GIS, the detection of spatio-temporal LULC dynamics has become easy, quick and cost-effective (Avelar et al., 2009, Santos et al., 2015). Remote sensing is considered the most efficient technology to handle these problems since it can explicitly reveal spatial patterns of land cover change over a large geographic area in a regular and consistent way (Lillesand and Kiefer, 2000; Singh et al., 2012). These advantages have attracted great interest in the scientific community. Moreover, the rich archive and spectral resolution of satellite images are the most important reasons for their use (Lillesand and Kiefer, 2000; Gillanders et al., 2008). Thus, change detection has become a major application of remotely-sensed data because of repetitive coverage at short time intervals, which is useful for tracking changes in LULC over longer periods of time and at more varied temporal scales than what is typically done with field experiments or ground inventory (Weng, 2002; Fan et al., 2007). GIS technology can be used to further analyze LULC datasets for different applications (e.g. Xiao et al. 2006; Avelar et al., 2009; Belal and Moghanm, 2011; Sylla et al., 2012; Obade and Lal, 2013; Nguyen et al., 2015).

Various techniques are available to extract meaningful information of LULC types from remotely captured datasets. In the past, most LULC classifications have been created using a pixel-based analysis of remotely sensed imagery. They used either unsupervised (K-means and ISODATA) classification, supervised (maximum likelihood) classification and some combination of which the most commonly used classification technique is the supervised classification technique (e.g. Gautam and Chennaiah, 1985; Bewket, 2002; Mas et al., 2004; Dessie and Kleman, 2007). Recently, object-based image analysis has been applied more frequently for remote sensing image classification than pixel-based analysis (Blaschke, 2010; Whiteside et al., 2011). Pixel-based methods classify individual pixels mainly using spectral patterns. The use of spatial or contextual information from neighborhood pixels remains a critical drawback to pixel-based image processing (Luo et al., 2011). On the other hand, object-based methods allow integration of different object features, such as spectral values, shape and texture (Baatz and Shape, 2000; Manakos et al., 2000; Definiens, 2009). One of its strength is the ability to combine spectral information and spatial information for extracting target objects (Baatz and Schäpe, 2000; Whiteside et al., 2011). However, accuracies of object-based approach differ depending on the nature of landscape and type of images used for analysis (Dronova et al., 2011).

In light of the existing knowledge it can be concluded that LULC study should be location specific and continuous. There is also a need of choosing appropriate datasets and techniques in order to extract meaningful LULC information with the highest possible accuracy.

2.2 Geo-informatics for understanding drivers of LULC changes

LULC changes in different parts of the world have been propelled by a set of driving forces that determine the trajectories of landscape development (Serneels and Lambin, 2001; Bürgi et al., 2004; Campbell et al., 2005; Beilin et al., 2014). Many, if not most, LULC changes are intended or unintended consequences of human decisions and the subsequent actions. The drivers of these changes are broadly categorized as social, economic, environmental, policy/institutional and technological factors (Lambin et al., 2003).

The social drivers are related to the changes in LULC as a result of population growth (Bewket, 2002; Brink et al., 2014). The number of people currently residing on Earth is widely acknowledged as an important variable in influencing the condition of ecosystems, e.g. LULC types and their changes (Meyer and Turner, 1992; Braimoh, 2004). There is also a growing recognition that how population is distributed across age groups, urban and rural regions, living arrangements, working conditions, and geographic regions affects consumption patterns and, therefore, impacts on LULC distributions (Beilin et al., 2014). The economic drivers are the consequences of human impact on LULC types to improve the quality of life, the outputs of which are determined by the natural resources (Wood et al., 2004). Land use practices, such as fuelwood collection, livestock grazing, charcoal making and road expansion, are examples of economic drivers that can cause LULC changes (Nagendra et al., 2003; Dessie and Kleman, 2007; Njenga et al., 2013). Furthermore, agricultural expansion is also recognized as the economic driver of LULC changes (Woodhous, 2012).

The environmental drivers include the characteristics and processes of the natural environment or climate induced changes, such as weather conditions (rainfall variability, moisture availability and temperature), topography (slope and altitude), quality of land (erosion), fire events and flooding (Thiombiano and Tourino-Soto, 2007; Yang et al., 2008; Martinez et al., 2011; Kicklighter et al., 2014; Román-Cuesta et al., 2014). Variations in those natural conditions have influences in determining the LULC changes in different landscape. The policy/institutional drivers are those factors, such as land tenure and legislation, that influence and lead to LULC changes (McCann, 1995). Technological factors also condition the LULC changes by influencing those who depend on land. Developments of technologies (e.g. seed varieties, farm tools and mechanization) have profound influence and triggered the dynamics of LULCs in a given landscape (Dietrich et al., 2014). Understanding these factors that cause changes in LULC is essential for predicting future changes that are likely to occur using models (Serneels and Lambin, 2001) or development of management strategies and policies to ameliorate or prevent further decline of natural resources (Tekle and Hedlund, 2001; Mottet et al., 2006). Thus, in recent decades, the study of drivers of LULC changes has become one of the major goals of global change research (Lambin and Geist, 2006).

Several studies were devoted to investigate drivers of LULC changes using different methodologies, including statistical analyses, case studies, geo-informatics-based spatial statistical models and simulations (Bewket, 2002; Serneels and Lambin, 2001; Beilin et al., 2014). Although the drivers of changes are recognized as important, and the literature is growing in both size and sophistication, few studies integrate both social and ecological data from case studies to examine the links between the drivers and their effect on LULC dynamics. Yet, the trends in LULC dynamics reveal concern globally, but driven markedly by different reasons or causes in various landscapes (Bewket, 2002; Dessie and Kleman, 2007; Beilin et al., 2014).

While considerable efforts have been made to identify the drivers, findings from different studies suggest that the drivers of LULC changes vary from place to place depending on location-specific factors. Many places of the world have also experienced multiple drivers of land use change simultaneously, and the impacts of each depend in part on how different drivers of LULC change interact (Campbell et al., 2005; Kicklighter et al., 2014). They are sometimes remote in space or time from the observed changes, and frequently involve macro-economic and policy changes that are difficult to anticipate (Serneels and Lambin et al., 2001). In addition, there is a significant debate on the level of influence by drivers of changes, making generalization nearly impossible (Bewket, 2002; Tegene, 2002; Dessie and Kleman, 2007).

Thus, there is a clearly need for empirical investigation into the problem. In particular, integrated analyses of drivers on LULC dynamics is limited. As stated by Beilin et al. (2014), the drivers of changes are still contentious issues and further research is necessary. Earlier studies (Serneels and Lambin, 2001; Lambin et al., 2003; Mottet et al., 2006) also underlined the need for detailed understanding of drivers that can enhance the ability to project future outcomes and intervention options. Such a comprehensive study of drivers would be useful to better understand the interrelationships between local people and the land resources, i.e. LULC types, which can help as a basis for the development of more appropriate and sustainable land use systems. Hence, it can be summarized that any intervention to address drivers of changes properly and development of sustainable landscape ought to begin with empirically supported and locally-specific understanding of the multiple drivers affecting the LULC types.

2.3 Geo-informatics for quantifying ecosystem services in relation to LULC changes

Scientists have described society's critical dependence on ecosystem services (Westman, 1977; MEA, 2005; Bateman et al., 2013; Knoke et al., 2016). Ecosystem services are the direct and indirect contributions of ecosystems to human wellbeing and survival (MEA, 2005; Schägner et al., 2013; Fisher et al., 2009; Costanza et al., 2014). They result from interactions between biotic and abiotic components of ecosystems (Singh, 2002). The ecosystem services include not only provisioning (e.g. food production, raw material and water supply), but also regulating (e.g. climate regelation, water purification and disturbance regulation), supporting (e.g. nutrient cycling, pollination and soil formation) and cultural services (e.g. aesthetic values) (MEA, 2005). Because of their relevance to society, these ecosystem services as well as their economic values have become focuses of interest over the last decade (Troy and Wilson, 2006; Butler et al., 2013) and among one of the popular issues in ecological economics (Troy and Wilson, 2006; Bateman et al., 2013).

The ecosystem service values (ESVs) are directly correlated to the situation of ecosystems, e.g. LULC types, in a given area (de Groot et al., 2002; Styers et al., 2010). The forests, as one of the LULC types, offer important ecosystem services, such as erosion protection, water supply and quality (Allen, 2004) apart from provisioning services (Sukara, 2014). They reduce the risk of soil erosion when they are found on steep slopes (Zeleke and Hurni, 2001), and in higher elevations, usually increase local water supply by collecting moisture out of the fog-saturated atmosphere that would otherwise remain in vapor form (Martínez et al., 2009). They also provide long-term storage of climate-altering greenhouse gases (Manrique et al., 2011). Besides storing greenhouse gases and capturing water, it has also been demonstrated that they provide the highest quality stream water and as natural filters against pollutants, and, thus, play a key role in water quality (de Souza et al., 2013; Fiquepron et al., 2013). Thus, instead of viewing nature conservation as a trade-off for human well-being, it is now also looked up on as an important contributor (MEA, 2005).

Dynamics of LULC can cause changes in the values of ecosystem services (Kreuter et al., 2001; Hu et al., 2008; Polasky et al., 2011). It may increase the provision of some services while decreasing others that affect the ability of biological systems to support human needs, indicating ecological degradation (Polasky et al., 2011), or vice versa. As changes in ESVs differ depending on the direction and/or magnitude of the LULC dynamics, most of the available studies were location-specific. For instance, Leh et al. (2013) revealed a general decline of ESVs while Wang et al. (2014) found the opposite, i.e. an increasing trend. Consequently, making direct use of such results to other areas might lead to erroneous conclusions. Nevertheless, because of population growth, economic pressure, and urban growth, many natural ecosystems are continuously being altered, destroyed or transformed, especially during the last decades (Dessie and Kleman, 2007; Martínez et al., 2009; Biazin and Sterk, 2013). Such ecosystem degradation threatened a continued supply of ecosystem services, while, at the same time, the demand for ecosystem services are increasing with human population growth (MEA, 2005; Guo et al., 2010). Globally, the Millennium Ecosystem Assessment documented that approximately 60% of the ecosystem services are being degraded or used unsustainably, including wood, fresh water, air and water purification, and the regulation of regional and local climate and natural hazards (MEA, 2005).

It is possible to value the services delivered by ecosystems and analyze their changes by using biophysical models. Valuation concerns the assessment, appraisal or measurement of the importance or values of ecosystem services as foundations of human societies (de Groot et al., 2002). The quantification and analyses for changes of ESVs serve different purposes, including as important tools to raise awareness (Liu et al., 2010), contribute to developing knowledge on management of natural capital (Costanza et al., 1997; Frélichová et al., 2014), improve decision making for allocation of scarce resources among competing demands (Guo et al., 2001; Barral and Oscar, 2012), formulate polices (Schägner et al., 2013) and provide a stimulus to conserve the ecosystems that offer the most valuable services (Konarska et al., 2002; Bateman et al., 2013). As a result, interest in ESVs has evolved rapidly in both the scientific communities and policy makers (Turner et al., 2003; Troy and Wilson, 2006; Butler et al., 2013; Costanza et al., 2014). The approach is widely utilized as a framework to understand and analyze the relationships between societies and ecosystems.

Studies on quantifying the ESVs and analyzing their changes has received wide attention as one of the most significant and fastest evolving areas of research in environmental and ecological economics after the publication of Costanza et al. (1997), who proposed a list of ecosystem service value coefficients of biomes (LULC types) based on synthesis of previous studies and estimates of global ESVs. Although the proposed global value coefficients have been criticized because of uncertainties (Limburg et al., 2002; Hein et al., 2006; Maes et al., 2012), a number of researchers working in regions where data are scarce have used them through benefit transfer method and paved the path to the science of ecosystem service valuation (Kreuter et al., 2001; Zhao et al., 2004; Wang et al., 2006; Li et al., 2007; Hu et al., 2008; Kubiszewski et al., 2013). The benefit transfer method refers to the process of using existing values and other information from the original study site to estimate ESVs of other similar location in the absence of site-specific valuation information (Kreuter et al., 2001; Kubiszewski et al., 2013).

The growing body of literature on the valuation of ecosystem services includes studies on changes on ESVs (Kreuter et al., 2001), analyses of the effect of spatial scales on the valuation of ecosystem services (Hein et al., 2006), land use planning based on ecosystem service assessment (Barral et al., 2012), quantifying and mapping of multiple ecosystem service changes (Leh et al., 2013), bringing ecosystem services into economic decision making (Bateman et al., 2013), and assessment of values of ecosystem services in nature reserve (Wang et al., 2014). Though many case studies on ESVs have been carried out, too few have paid enough attention to long-term changes of ESVs in response to LULC changes and adjusting available coefficients into a higher local validity during estimation of ESVs. Little attention has also been focused on the spatial visualization and mapping results of ESVs and their changes (Maes et al., 2012; Leh et al., 2013). Most of previous economic valuations have been non-spatial when estimating and describing the values with statistical data (Chopra, 1993; Hope and Maul, 1996; Higgins et al., 1997).

In recent years, remote sensing and GIS technologies were commonly applied in most of the studies during the spatially explicit ecosystem service estimation processes. The former offers opportunities of generating LULC types for a given area that can be utilized as proxies of measurements while the latter is used for estimating and mapping their distributions (Konarska et al., 2002; Zhao et al., 2004; Wang et al., 2006). Ecosystems and their services are spatially explicit, and this makes GIS very appropriate for the analyses. As primary datasets are expensive, or sometimes scarce in some regions, secondary data consisting of spatial units, such as LULC classes, are also, more often, used as proxies for estimation

(Kreuter et al., 2001; Maes et al., 2012). In addition, for the corresponding value coefficients, The Economics of Ecosystems and Biodiversity (TEEB) valuation database was also developed, mainly, based on literature of case studies in different parts of the globe (van der Ploeg and de Groot, 2010).

In summary, as changes in ESVs differ depending on the condition of LULC types, site specific study is needed to understand the relationship between societies and ecosystems. In addition, the variation of changes in ESV through more locally valid coefficients, when compared to the often used global coefficients (Costanza et al., 1997), and impact of the variation of the value coefficients over time have not been investigated.

2.4 Geo-informatics for scenario modelling of LULC changes

Development of effective land use planning strategies for sustainable resource use and conservation requires knowledge of future LULC patterns and changes (Serneels and Lambin, 2001; Bhattacharjee and Ghosh, 2015). One way of exploring the potential future situation of land resources is through the use of modelling. Hundreds of LULC change models have been described in the literature on landscape ecology, geography, urban planning, economics, regional science, computer science, statistics, geographic information science, and other fields (Brown et al., 2004). Models can be categorized according to the amount of information they contain, namely whole landscape models, distributional landscape model, or spatial landscape models (Baker 1989).

Among the categories of models in view of functional and methodological aspects, GIS-based modelling is nowadays a frequently and widely used approach in LULC research (Paudel and Yuan, 2012). Since the 1980s, the development of GIS has opened new horizons as essential technology for the management and manipulation of spatial data sets (Burrough, 1986; Carver et al., 2012; Nguyen et al., 2015). From routinely performing work-related tasks to scientifically exploring the complexities of our world, GIS offers the geographic advantage to become more productive, aware and responsive citizens of the planet Earth (Esri, 2011). Specifically, GIS is beneficial in LULC modelling because it is able to provide visual-based simulation environment, data management and well developed algorithms to deal with datasets of high spatial detail with information content. It is valuable in determining the spatial resolution necessary for LULC modelling and enabling organizations to leverage data to make more informed decisions on all fronts. GIS has also become a key part of scenario modelling studies because it has proved useful in LULC modelling processes, such as the spatial and temporal distribution of inputs and parameters controlling the changes, i.e. the drivers behind the process (Nourqolipour et al., 2015). Additionally, GIS provides such extensive opportunities to model LULC dynamics in a spatially explicit manner (Qiang and Lam, 2015). Such nature and ability to accurately and spatially represent features has paved the attention of land use scientists (Esri, 2011; Paudel and Yuan, 2012; La Rosa, 2014; Latinopoulos and Kechagia, 2015).

Simulation models have been used by a large number of research groups to explore when and where future LULC changes would occur based on the goals of a particular study through integrated multidisciplinary research of complex environmental processes and their interactions (Wu et al., 2006; Schaldach et al., 2011; Grinblat et al., 2015; Nourqolipour et al., 2015). There are differences in modelling approaches of these studies, which often relate to differences in the purpose of the study. For example, Wu et al. (2006) predicted land use change in Beijing, China. Grinblat et al. (2015) developed and simulated dynamics of agricultural land use in Mali, West Africa. Nourqolipour et al. (2015) employed a GIS-based model to analyze spatial and temporal development of palm oil plantations in Kuala Langat district, Malaysia. These results showed wide variety of information about future LULC types in each of the studied area. Since the drivers of LULC change differ from place to place, it is important to understand their location-specific interaction and reasonably predict the future demand of land, which is the key in land use planning and management specific to the areas under investigation (Wardell et al., 2003; Kindu et al., 2015; Nourqolipour et al., 2015).

In combination with the increasing tendency to future-orientated multidisciplinary research and availability of numerical impact assessment techniques (e.g. Serneels and Lambin, 2001; Grinblat et al., 2015), spatially explicit simulation modelling has become an attractive approach to assess and visualize future LULC patterns and changes under set of scenarios. Such approach provide good information on how various courses of action may affect the future of a given resource in which today's decision might be played out (Sun et al., 2012; Martinuzzi et al., 2015). Additionally, it gives a chance to estimate the changes of ESVs in response to LULC dynamics (Hu et al., 2008; Dallimer et al., 2015). In this case, it can enrich our understanding of human activities about resource use and conservation (Bachelet et al., 2003). In particular, knowing the potential outcomes of alternative scenarios can be a powerful tool when making and implementing difficult policy decisions (Sun et al., 2012).

Thus, it significantly contributes to the understanding of the potential constraints and opportunities associated with various course of actions and enhances early decision making process to minimize consequences and mitigate harmful impacts. Therefore, the simulation outputs could play an important role in facilitating the identification and planning of management strategies that could reverse the trend and promote improved management of natural resources (Bhattacharjee and Ghosh, 2015).

Based on the existing knowledge, it can be concluded that several LULC simulation models are available. However, regardless of current advances in modelling, they are limited in their ability to simulate using specific datasets and contexts within which they are calibrated. While they are useful for identifying future pattern and changes of LULC types within a particular framework and projecting trends defined by the data, these models are limited in their ability to generate results or predict in the future situations where the drivers are different from those operating in the real situation. Thus, research on modelling LULC patterns and changes should be based on level of complexity of the target area, scale of application, and most importantly locally valid datasets and their associated drivers. Owning to the complex dynamics to the LULC types in a given landscape, simulating the entire landscape is also advisable for inclusive recommendations instead of working on models that only examine a specific part of LULC types, such as modelling crop or grassland ecosystems independently (Sabatier et al., 2010; Wang et al., 2014).

3 Materials and methods

3.1 Study area

3.1.1 Location, topography and demography

The study was conducted in the Munessa-Shashemene landscape of the Ethiopian highlands, which is located within Munessa and Arsi-Negele Districts (between 7°20'01.23" to 7°35'13.3" N and 38°39'43.3" to 38°59'57.31" E) about 200 km south of Addis Ababa (Figure 3.1).

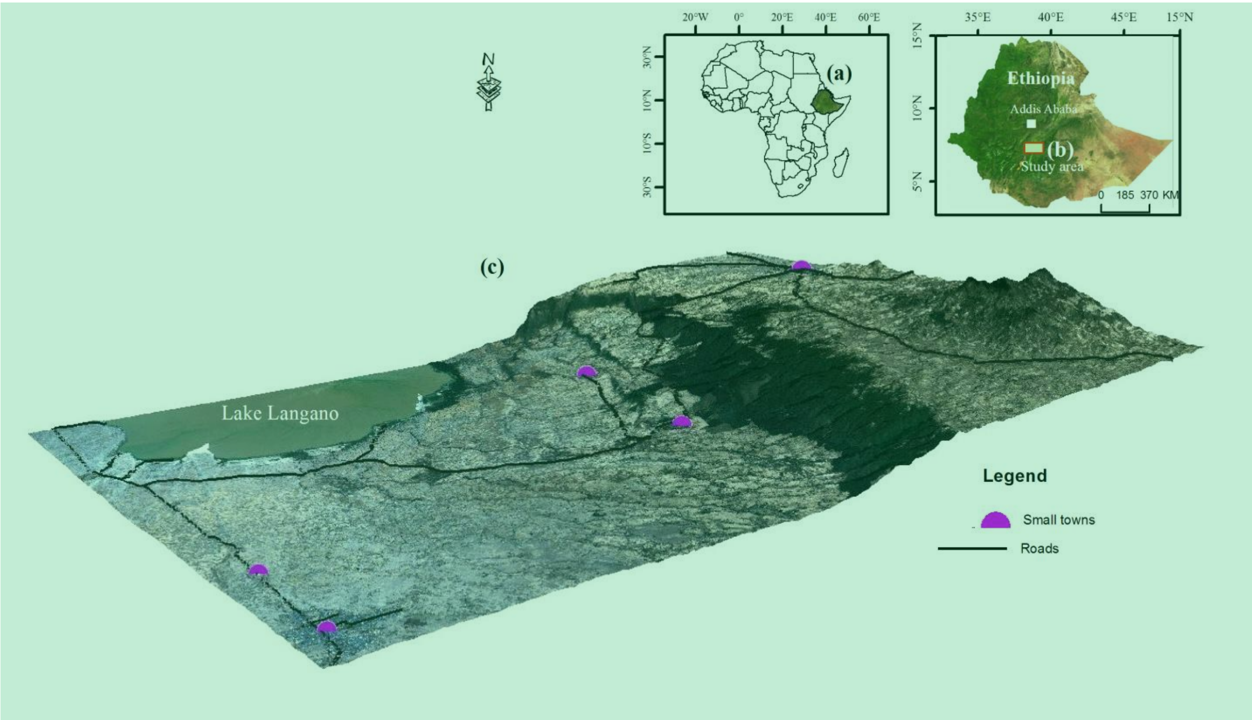

Figure 3.1: Location of the study area: (**a**) African position, (**b**) Ethiopian context, (**c**) 3D view of study landscape. Study landscape background is RapidEye image of year 2012 with RGB (red, green and blue): bands 3, 2 and 1, draped on Aster Digital Elevation Model (DEM) with three-fold vertical exaggeration.

The Munessa-Shashemene landscape is characterized by diverse topographic conditions with a total area of about 1,091 km^2. The elevation ranges from 1,500 m at the Central Rift Valley lakes to over 3,400 m at the Arsi-Bale massif in the eastern part of the study landscape. A mountainous and undulating topography with steep slopes characterizes the eastern part of the landscape, and gentle to moderate slopes characterizes the western part.

The area is one of the highly populated parts of Ethiopia. The population of Munessa and Arsi-Negele Districts, based on the 2007 census result, was estimated at 430,728 (CSA, 2007). The majority the people (84%) reside in the rural areas.

3.1.2 Climate and soils

The climatic conditions in the highlands of Ethiopia are tropical highland monsoon where the seasonal rainfall distribution is controlled by the movement of the inter-tropical convergence zone and highly influenced by as result of complex topographic (altitudinal) differences (Taddese, 2001). There is a decline in mean annual temperature and increase in mean annual rainfall with increasing altitude. The climate of the Munessa-Shashamane landscape falls into three zones that include sub-humid, humid and cold highlands (Hurni, 1998). The rainfall has a bimodal distribution. The short and main rainy seasons occur from March–May and July–September, respectively. Meteorological station records show that the annual rainfall is about 1,200 mm at Degaga town (2,000 m), which is found in the study area. Mean annual temperature is 15 °C.

The soils of the area are closely related to their parent materials and their degree of weathering. They are rich in clay and classified as Mollic Nitisols, Vertisols (Mazic Vertisols), Umbric Andosols, Humic Ubmrisols, Mollic Cambisols and Alisols (Fritzsche et al., 2007). The first three cover large area of the landscape and agriculturally important soil types. The Nitisols are reddish in color and have moderate CEC, relatively high organic matter content and total nitrogen. The Vertisols are dark and heavy clay soils and occupy waterlogged plains. Andosols are also dark in color and have high water holding capacity (Aran et al., 2001).

3.1.3 Agriculture and biodiversity

Farmers in the area are engaged in agriculture with a mixed farming system that is carried out on a subsistence scale, hence, livestock rearing and crop cultivation is common in all the altitudinal ranges (Lemenih et al., 2005). The livestock kept in the study area are, mainly, cattle and goats, and reared in a free grazing system. They provide the draught power for the farming operation as well as for generation of income for the households. A variety of crops are produced by a household under rain-fed system, mainly, with one harvest per year. Major crops grown are barley (*Hordeum vulgare* L.), wheat (*Triticum aestivum* L.), maize (*Zea mays* L.), beans (*Phaseolus vulgaris* L.), sorghum [*Sorghum bicolor* (L.) Moench.] and teff [*Eragrostis tef* (Zucc.) Trotter.].

Apart from croplands, the landscape also comprises mosaics of other land use/land cover (LULC) types, mainly natural forests, grasslands, plantation forests, woodlands, settlements and water bodies. The natural forests belongs to the tropical dry Afromontane forest (Teketay and Granström, 1995). They are dominated by indigenous tree species, namely *Podocarpus falcatus* (Thunb.) R. B. ex. Mirb., *Croton macrostachyus* Del, *Ekebergia capensis* Sparrm., *Celtis Africana* Burm.*f*., *Prunus Africana* (Hook.*f*.) Kalkm., *Polyscias fulva* (Hiern) Harms, *Pouteria adolfi-friederici* (Engl.) A. Meeuse [*synonym: Aningeria adolfi-friederici* (Engl.) Robyns & Gilbert], *Syzygium guineense* (Willd.) DC. (Tesfaye et al., 2010). The woodlands are dominated by *Acacia* species and found in the lower altitude of the study landscape. The plantation forests are composed of exotic species, mainly *Cupressus lusitanica* Miller, *Pinus patula* Schlechtendal & Chamisso and *Eucalyptus* spp (Chaffey, 1980) established starting from the late 1960s (Lundgren, 1971). The water body (Lake Langano) receives most of its water from small rivers and streams that drain from the Arsi Mountains and is rich with biodiversity.

Lake Langano contains five species of barbus fish, namely *Barbus paludinosus*, *Garra dembecha*, *Labeobarbus intermedius*, *Clarias gariepinus* and *Oreochromis niloticus* (Vijverberg et al., 2012). The lake has an estimated fish production of about 1,000 ton year^{-1}, which is about 7% of the total annual average catch of the country for the period from 1999 to 2009 (Tesfaye and Wolff, 2014). The surrounding of the lake is endowed with *Acacia* species dominated vegetation, which is a home for many bird species. Resort areas are booming since the lake has extended beaches, and the water is more suitable for swimming, sunbathing, camping, water sports and bird-watching (Estifanos, 2008; Kidane-Mariam, 2015).

3.2 Datasets

3.2.1 Satellite imagery

Landsat and RapidEye imagery were the core datasets for classification of major land use/land cover (LULC) types of the study landscape (Kindu et al., 2013, see Publication I in the Appendix). The Landsat imagery data included Landsat MSS, Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM+) scenes of the year 1973, 1986 and 2000, respectively. These datasets were acquired from the National Aeronautics and Space Administration (NASA) through their EOS Data Gateway Database (Table 3.1). The RapidEye imagery data were used for the study year 2012. The images were obtained from RapidEye Science Archive (RESA) supported by German Aerospace Center (DLR). They were ortho-rectified level 3A and geometrically corrected. Both Landsat and RapidEye imageries were acquired in the same season. Images of the same season were selected to reduce the effect of seasonal discrepancies on the classification results (Lu et al., 2004).

A 30 m Digital Elevation Model (DEM), based on Aster imagery, was also employed in order to drive slope gradient and agro-ecological zones of the study landscape. In addition, ancillary data were utilized during analysis, including topographic maps, field data, thematic layers (roads and towns), *Kebele*[1] (Village) and District (Woreda[2]) boundaries. All datasets were projected to the Universal Transverse Mercator (UTM) projection system zone 37N and datum of World Geodetic System 84 (WGS84), ensuring consistency between datasets during analyses. The pre-processing of datasets was made using ArcGIS, ENVI and PCI geomatics software packages.

3.2.2 Other datasets

Other datasets required for the study were generated through both primary and secondary sources, including reports, formal household survey, group discussion with elders, field observations, measurements, informal discussion with individual farmers and development agents (DAs) working at the study landscape. Two phases of field work were conducted. Both primary and secondary data collection and detailed household surveys, using semi-structured questionnaires, and group discussions were conducted between November 2011 and January

[1] *Kebele is* the smallest and lowest level of administration units in Ethiopia
[2] Woreda is the local administrative unit above *Kebele*

2012. GPS coordinates of target LULC types and market centers were collected, and information regarding each site was noted. Further field work for additional secondary data was conducted during January-February, 2013.

The secondary datasets were mainly collected and synthesized from published literature, official statistics, and policy documents. Population data were obtained from the Central Statistical Authority (population of 1994 and 2007) and Woreda Offices (population of 2012) of the study landscape. The road network datasets were derived in ArcGIS environment from multiple sources, including Central Statistical Authority of Ethiopia, topo map, aerial photographs and RapidEye images. The rainfall datasets were obtained from the Ethiopian National Meteorological Service Agency. Station records from 1981 to 2008 of Langano located in the lower part of the study landscape were used. The long term annual average rainfall (1950 to 2000) from the WorldClim database produced by Hijmans et al. (2005) and available online were also utilized.

Table 3.1: Summary of datasets used in this study.

Datasets	Acquisition date (s): Day/Month/Year	Pixel Resolution/Scale	Source (s)
Satellite datasets			
Landsat MSS	30/1/1973	60 m	NASA (http://glovis.usgs.gov/)
Landsat TM 5	21/1/1986	30 m	NASA (http://glovis.usgs.gov/)
Landsat ETM+	5/2/2000	30 m	NASA (http://glovis.usgs.gov/)
RapidEye	1/2/2012	5 m	RESA (http://eyefind.rapideye.com/)
Aster DEM		30 m	NASA (http://glovis.usgs.gov/)
Other secondary and primary datasets			
Aerial photographs (Black and white)	1/2/1972	1:50000	EMA
	1/2/1986	1:20000	EMA
Topo map		1:50000	EMA
Field data	10/2011–1/2012; 1/ 2013–2/2013		Field survey
Population data			CSA, District offices
Rainfall data			NMA, WorldClim database
Temperature data			NMA
Roads and towns			
Kebele boundary			
District boundary			
ESV coefficients			Costanza et al., 1997; vand der Ploeg and de Groot, 2010; Knoke et al., 2011
Price index data			http://data.imf.org/; http://faostat.fao.org/
Synthesized data			From report, articles, policy briefs

CSA= Central Statistical Authority; DEM = Digital Elevation Model; ESV = Ecosystem Service Value; NMA = National Meteorological Agency of Ethiopia; EMA = Ethiopian Mapping Agency; NASA = National Aeronautics and Space Administration; RESA = RapidEye Science Archive; MSS = Multispectral Scanner System; TM = Thematic Mapper; ETM+ = Enhanced Thematic Mapper

3.3 Methodology

3.3.1 Land use/land cover (LULC) classification and change analyses

Various techniques are available to extract meaningful information of land use/land cover (LULC) types from remotely captured datasets. Object-based image analysis (OBIA) was employed for the classification of major LULCs in the study landscape. This is owing to its ability to combine spectral information and spatial information for extracting target objects (Baatz and Schäpe, 2000; Whiteside et al., 2011). Figure 3.2 shows the methodological approaches employed to classify satellite images and analyze changes in LULCs. Detailed descriptions of the analysis methods are given in Kindu et al. (2013), see Publication I in the Appendix) and the overall approach is briefly explained in the following subsections.

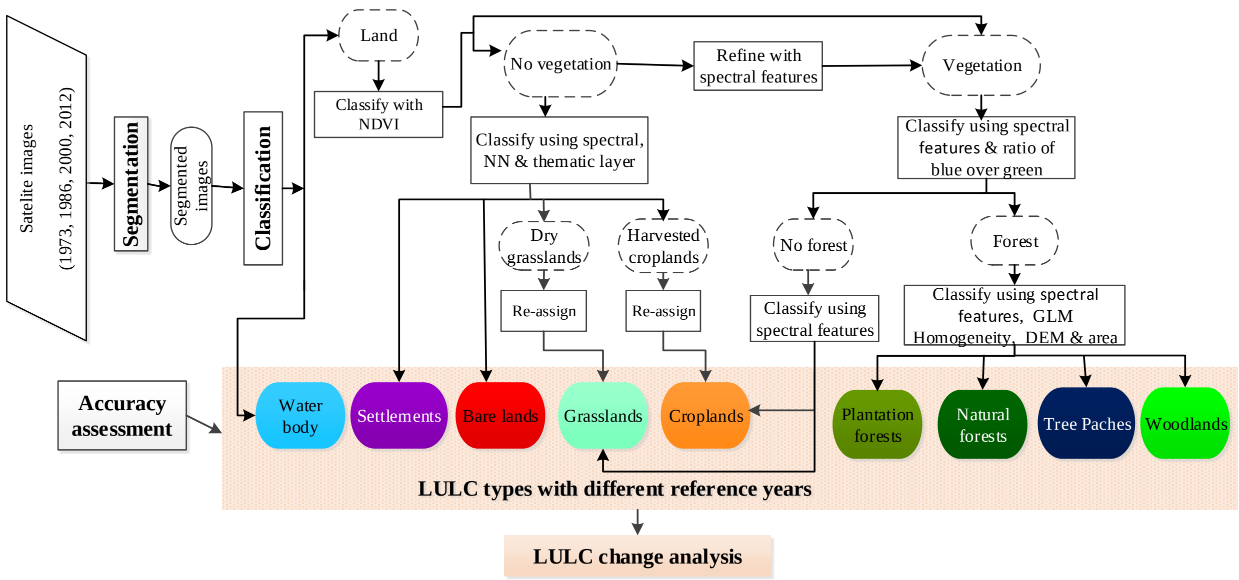

Figure 3.2: Methodological approach used for land use/land cover (LULC) types classification and analyze the changes in the study landscape. Dash line boxes are intermediate/temporal classes and colored boxes are final LULC classes.

3.3.1.1 Image segmentation

Image segmentation is an established way for the creation of objects or separated regions in an image required for successful object-based image analysis (Blaschke, 2010). There are many image segmentation algorithms developed depending on the type of specific tasks. In this research, the sophisticated segmentation algorithm, known as multi-resolution segmentation (MS), which is based on the Fractal Net Evolution Approach (FNEA) (Baatz and Schäpe, 2000) and available in eCognition Developer 8.0 software, was utilized. The MS algorithm is also an optimization procedure that minimizes the average heterogeneity for a given number of objects and maximizes their homogeneity based on defined parameters. These parameters[3], namely scale, shape, and compactness, are defined through trial and error to successfully segment objects in an image (Flanders et al., 2003; Yan et al., 2006; Tian and Chen, 2007; Definiens, 2009). Scale parameters, ranging from 8 to 500 with three different levels depending on the type of images, were used for the analyses (Table 3.2). The images were segmented in to three levels to facilitate the object-based classification depending on the nature of LULC classes to be detected. For instance, level 1 was used to handle those big size classes like water bodies, whereas level 3 was used for small size classes like tree patches.

Table 3.2: Parameters used for different images in each segmentation level.

Data Type	Resolution	Parameters Used in Different Segmentation Levels								
		Parameters for Level 1			Parameters for Level 2			Parameters for Level 3		
		Sc	Sh	Cm	Sc	Sh	Cm	Sc	Sh	Cm
Landsat MSS	60 m	40	0.2	0.8	15	0.2	0.8	8	0.2	0.8
Landsat TM 5	30 m	50	0.1	0.5	20	0.1	0.5	10	0.1	0.5
Landsat ETM+	30 m	50	0.1	0.5	20	0.1	0.5	10	0.1	0.5
RapidEye	5 m	500	0.2	0.7	200	0.2	0.7	50	0.2	0.7

Sc = Scale; Sh = Shape; Cm= Compactness

[3] Scale parameter identifies the highest heterogeneity allowed for the objects; Shape parameter balances spectral homogeneity versus shape of objects on segmentation outcome; Compactness parameter determines image objects based on their relative shape.

3.3.1.2 Object-based classification

The object-based classification was applied to a segmented image in order to assign a class to each of the segments using identified target LULC classes (Table 3.3). This was carried out in eCognition 8.0 software (Definiens, 2009). There are two approaches in eCognition to assign classes to segmented objects, which are membership functions and the nearest neighbour (NN) classifier. The membership function classifier uses the user's expert knowledge to define rules and constraints in the membership function from object features to control the classification procedure. On the other hand, NN classifier uses a defined feature space, e.g., using original bands or customized bands, and a set of samples that represent different classes in order to assign class values to segmented objects (Whiteside et al., 2011). Whenever applicable, both approaches were used during the classification process.

Table 3.3: Description of land use/land cover (LULC) types.

LULC Types	**General Description**
Bare lands	Areas of land those already get bad either due to erosion or misuses especially over grazing and crop cultivation.
Grasslands	All areas covered with natural grasses and small shrubs dominated by grasses.
Water body	Permanent lakes and other intermittent ponds.
Settlements	Build-ups (houses) in both urban and rural parts.
Croplands	Areas of land prepared for growing agricultural crops. This category includes areas currently under crop, and land under preparation.
Tree patches	Areas covered with scattered trees, bushes and shrubs along the study landscape. Small patches of forests (<0.5 ha) are also included in this category.
Plantation forests	Areas covered by man-made trees with minimum size of 0.5 ha.
Natural forests	Areas dominated by natural high forests, which are evergreen or deciduous.
Woodlands	Forests found below 1900 m (Teketay, 2000). Mainly dominated by *Acacia* spp.

The implemented classification scheme was hierarchical with three levels applied using a ‘‘top-down’’ approach. That is, the classification started from very general classes (level 1), which were further subdivided into more specific classes (level 2 and 3). The whole study landscape was broadly classified in level 1 into water body and land classes by using the spectral features from the mean value of objects in near infrared band. The second and third levels were used to extract the remaining target LULC types from the class land (Figure 3.2).

The object-based classification of target classes was achieved by using mainly thresholds of mean and/or standard deviation of spectral features (original bands of blue, green, red, red edge, and near infrared), customized bands (ratio of blue over green), thematic layers, DEM values, texture value of grey-level co-occurrence matrix (GLCM) homogeneity, and normalized difference vegetation index (NDVI). The NDVI was calculated using equation 3.1 (Reed et al., 1994).

$$NDVI = \frac{NIR - RED}{NIR + RED} \tag{3.1}$$

where NIR and RED = reflectance in the near infrared and the red bands, respectively.

The calculated NDVI values were used to further classify the class land into vegetation and no-vegetation classes. As the images were taken during the dry season, some dried vegetation with low NDVI was classified as no-vegetation class. Expert knowledge based developed rule set using red and red edge mean values were utilized to refine such classes from no-vegetation to vegetation classes. The vegetation class was again subdivided into forest and no-forest classes using the mean value of objects in red band and a value from blue over green ratio. The forest class was again classified to achieve the final target LULC types, namely plantation forests, tree patches, natural forests and woodlands. The standard deviation value of objects in red band, texture value of GLCM homogeneity, DEM and size (area) of the class objects were utilized to separate these four classes. On the other hand, the mean value of NIR was utilized to separate grasslands and croplands from the class no-forest. Based on the training samples, the NN classifier was also employed to further classify the no-vegetation class into settlements, bare lands, harvested croplands and dry grasslands. Again, associated to the dry season images acquisition, there were areas that were signed as bare lands although they were dry grasslands and harvested croplands. To avoid such confusion, the two classes (harvested croplands and dry grasslands) were temporally created under no-vegetation class and at the end, they were re-assigned to the class croplands and grasslands, respectively (Figure 3.2).

3.3.1.3 Accuracy assessment

Accuracy assessment determines the quality of the map extracted from remotely sensed data (Foody, 2002; Congalton and Green, 2009). The accuracy assessments were performed for classified images of 1973, 1986, 2000, and 2012. Considering the number of LULC classes, a minimum of about 40 random points were generated per class using stratified random sampling approach for efficient accuracy assessment (Congalton and Green, 2009). The corresponding reference class for each LULC type of the four reference years was collected from different data sources, including data from field visits, historical black and white aerial photos, topographic maps, and raw images. Raw images were used for those visually visible classes, e.g., forests and water bodies.

An error matrix or confusion matrix is a common practice employed for assessment of classification accuracy (Foody, 2002). The matrix compares information obtained by reference sites to that provided by classified image for a number of sample areas. Accordingly, overall accuracy, producer's and user's accuracies, and Kappa statistic were calculated from the error matrix.

3.3.1.4 LULC change analysis

Post-classification image comparison technique was employed to conduct change analysis (Singh, 1989). It was selected in order to minimize possible effects of atmospheric variations and sensor differences (Lu et al., 2004), but classification with high accuracy is a prerequisite for effective change detection (Foody, 2001). Independently classified images with the highest accuracy were used in the change detection process. Change statistics were computed by comparing image values of one data set with the corresponding values of the second data set in each period. This results in a summary table of the overall changes per class. The values were presented in terms of hectares and percentages. The percentage LULC changes were calculated using equation 3.2.

$$\text{Percentage LULC change} = \left(\frac{\text{Area}_{\text{final year}} - \text{Area}_{\text{initial year}}}{\text{Area}_{\text{initial year}}}\right)\times 100 \qquad (3.2)$$

where Area = extent of each LULC type. Positive values of equation 3.2 suggest an increase whereas negative values imply a decrease in extent. LULC conversion matrix between 1973 and 2012 was generated using ArcGIS 10 software and compiled in a matrix table, and the values were presented in terms of hectares.

Slope gradients and agro-ecological zones (AEZs) are conditional factors for LULC distributions and changes (Hurni, 1998; Hietel, 2004). The slope gradient and AEZs were developed from Aster DEM in ArcGIS environment (Kindu et al., 2013, see Publication I in the Appendix). By overlaying the classified maps of each reference year (1973, 1986, 2000 and 2012) on to the slope and AEZ map, thematic information showing relationship between LULC distribution and changes in each category was extracted.

3.3.2 Drivers of LULC changes

Understanding the multiple drivers of LULC changes requires a detail empirical and locally specific investigation into the problem. Thus, this specific research was undertaken by employing combination of techniques for data collection and analyses of the drivers of LULC change. The overall approach is briefly explained in this section.

3.3.2.1 Sampling and household survey

Selection of respondents to generate proper data from the household survey was carried out in a stratified method followed by a two-stage sampling technique. Accordingly, the study area was stratified in to three groups based on the developed AEZs, namely sub-humid highlands, humid highlands and cold highlands (Kindu et al., 2013, see Publication I in the Appendix). Then, in the first stage, a purposive sampling method was employed to identify representative *Kebeles* from each of the AEZs in the study landscape based on information collected from a reconnaissance survey. Thus, Gorbi Arba and Dagaga from the sub-humid highlands, Gujicha from the humid highlands and Koma Ocha from the cold highlands were selected. In the second stage, sample households were randomly selected from each *Kebele*. A total of 150 households were interviewed with a minimum of 30 from each *Kebele*. In addition, focus group discussions were conducted in each *Kebele*, mainly, with knowledgeable elders about historical and current LULC situation of the area. They were selected in consultation with *Kebele* development agents and the chairperson of each *Kebele*.

The survey questionnaires covered issues regarding socioeconomic characteristics of households, drivers of LULC changes, perception of the local people and ranking of the drivers. The selection of explanatory variables (drivers) of LULC changes incorporated in the questionnaire was based on literature and expert knowledge of the area. Similar types of issues were also covered during group discussions.

3.3.2.2 Data analyses of drivers of LULC changes

The generated datasets were analyzed using combinations of techniques, including descriptive statistics, GIS-based processing and regression analyses (Figure 3.3). Descriptive statistics of simple frequency analyses were used to describe socioeconomic characteristics of households and to summarize their responses and rankings of drivers of the land use/land cover changes. Data collected through group discussions, qualitative interview, observations and from the literature (e.g. reports and articles) were analyzed qualitatively. Association/differences in perceptions among respondents in different *Kebeles* concerning drivers of LULC changes were also investigated using non-parametric test, namely Pearson's Chi-square Test. Additionally, relationship between trends of population and croplands were explored (Kindu et al., 2015, see Publication II in the Appendix).

Standardized rainfall anomalies and coefficient of variations were employed to evaluate inter-seasonal rainfall fluctuations. Standardized anomalies of rainfall of the growing season were calculated using equation 3.4 and used to assess frequency of droughts as in Bewket and Conway (2007).

$$\mathrm{SRA} = \frac{(\mathrm{P}_t - \mathrm{P}_m)}{\sigma} \tag{3.3}$$

where SRA = standardized rainfall anomaly of growing season, P_t = growing season rainfall in year t, P_m = long-term mean rainfall of the growing season over a period of observation and σ = standard deviation of rainfall in growing season over the period of observation.

Farmers' perceptions of drivers for LULC changes are a function of household or socio-economic attributes (e.g. gender, age or occupation), which were generated during the survey of sample households (Table 3.4). A quantitative logistic regression analysis was employed at household level to identify the main socio-economic determinants of farmers' perception to some of the LULC drivers for the observed changes. The dependent variable, i.e. perception of a particular variable as a driver, is a dichotomous variable or a binary response that was generated from the questionnaire survey. On the other hand, the independent socio-economic variables are a mixture of discrete and continuous variables. Logistic regression analysis was a suitable statistical procedure to examine the relationship between the perception (dependent) and the various socioeconomic (independent) variables, since it is an effective technique for the analyses when the dependent variable is binary (Lesschen et al., 2005), which is the case in the present study. This logistic regression was also employed to identify the significant factors of LULC changes of the whole study landscape from 1973 to 2012 using pixels as unit of analysis (Verburg et al., 2004). The dependent variable of the landscape level study was the

LULC change/no change, whereas the independent variables were composed of social, economic and environmental factors.

Table 3.4: Characteristics of sample households in the study landscape (N = 150)

Household attributes	**Value**
Interview (gender: male, %)	83
Average household age (years)	48
Education (literate, %)	35
Household occupation (farming, %)	92
Mean household size (Number)	7
Mean land holding size (ha)	2.5
Mean household income (Birr*/year)	32787

* Ethiopian currency: at the time of the study, 1 USD = 17.48252 Birr.

In both cases, household and landscape level, the logistic regression function, which estimates the likelihood of the effects of the independent (explanatory) variables on the dependent (response) variable, is of the form (Lesschen et al., 2005):

$$\text{Logit}(Y) = \alpha + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_3 \ldots + \beta_n X_n \tag{3.4}$$

where Y = the dependent variable indicating the probability that Y =1, α = the intercept, β1……βn = coefficients of the associated independent variables, and X1…Xn = the independent variables. Before the analysis, the set of independent (predictor) variables were tested for multi-collinearity using a collinearity diagnostics index in linear regression analysis in SPSS (Rawlings et al. 1998).

The overall approach employed in this study for landscape level analysis of drivers (Fig. 3.3) involved comparison of LULC changes with the potential spatially explicit explanatory variables on a cell by cell basis (Serneels and Lambin, 2001; Verburg et al., 2004; Rutherford et al., 2007; Chen and Pontius, 2010) at 60 m spatial resolution. This scale was chosen because it is related to the unit of the lowest scale for 1973 landscape image from where the historical LULC data were generated. The datasets were prepared and changed to raster format in GIS environment at the same spatial extent and geographical coordinates. The raster datasets were, again, converted into ASCII format in GIS Environment, and each ASCII data representing each variable was changed into a column format using MATLAB to make them suitable for statistical analysis. At the end, the column datasets were analyzed in SPSS using binary logistic regression to identify the relations between a set of explanatory variables (drivers) and the location of changes in LULC at landscape level.

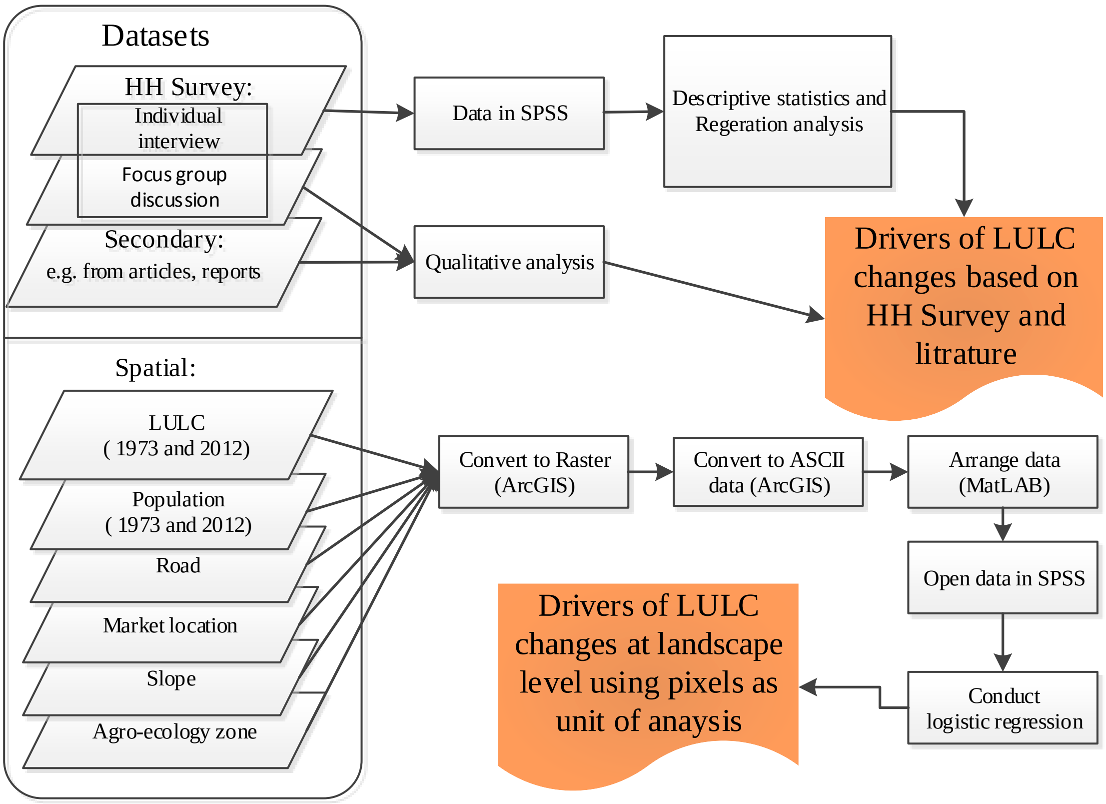

Figure 3.3: Methodological approach to investigate drivers of LULC changes using datasets from multiple sources.

3.3.3 Estimating ecosystem service values and their changes

The ecosystem service values (ESVs) and their changes were estimated using LULC datasets (Kindu et al., 2013, see Publication I in the Appendix) derived from remote sensing imagery as proxy of measurements with their corresponding value coefficients. The procedures followed for estimating ESVs and their changes are described below.

3.3.3.1 Ecosystem service value coefficients

Two types of ecosystem service value coefficients of the target LULC types were used for this study. The first were global coefficients adopted only from Costanza et al. (1997) ecosystem service value coefficients using representative biome as a proxy for each LULC type. They proposed and employed in their ecosystem service valuation model 17 types of ecosystem service value coefficients for 16 biomes (LULC types). In this study, each of the nine LULC types of the different reference years were compared with those representative biomes (LULC types) in order to obtain their corresponding ecosystem service value coefficients identified in Costanza et al. (1997). The second types of coefficients were more conservative and modified from those employed by Costanza et al. (1997) using expert knowledge of the study landscape conditions and other studies, mainly, from The Economics of Ecosystems and Biodiversity (TEEB) valuation database (van der Ploeg and de Groot, 2010) and Knoke et al. (2011). Detailed approach about modification and summary of conservative value coefficients can be found in Kindu et al. (2016, see Publication III in Appendix). Both approaches of directly using the available global value coefficients or their modification have been applied by a number of researchers for similar studies in data scarce areas (e.g. Kreuter et al., 2001; Zhao et al., 2004; Wang et al., 2006; Li et al., 2007; Hu et al., 2008; Kubiszewski et al., 2013). All the value coefficients were converted into 1994 US$ per hectare per year to facilitate the estimation process of ESV changes. The value coefficients were also adjusted using consumer price index and producer price index to check the effect of time development in the coefficients on the overall estimation of ESV changes. National average per capita gross domestic product (GDP) of 1973 and 2012, with the corresponding population of the studied landscape, was used to estimate GDP of the landscape.

3.3.3.2 ESV data analyses

The overall approach employed in this study involved estimating ESVs for 1973, 1986, 2000 and 2012 reference years, mapping their spatial distribution and computing the changes between study periods. The LULC datasets of each reference year used as a proxy for the measurement of ESVs were prepared, and the corresponding area (ha) was summarized in the GIS environment. In the ecosystem service estimation process, the value coefficients were assigned to each LULC type according to the value used by Costanza et al. (1997) and own modified coefficients. Then, the area of each LULC type was multiplied by its corresponding value coefficient to calculate the total ecosystem service value for a particular LULC type. The values for the LULC types in each reference year were summed up to estimate total ESV of the landscape for each reference year using equation 3.5.

$$ESV = \sum (A_k \times VC_k) \quad (3.5)$$

where ESV = total estimated ecosystem service value, A_k = the area (ha) and VC_k = the value coefficient (US$ ha^{-1} $year^{-1}$) for LULC type ‘k’.

The changes of ESV were obtained by calculating the difference between the estimated values in each reference year. ESV change statistics were computed by comparing values of one dataset with the corresponding value of the second dataset in each period. This resulted in a summary table of the overall changes in ESV. The values were presented as US$ and percentages. The percentage ESV changes were calculated using equation 3.6.

$$\text{Percentage ESV change} = \left(\frac{\text{ESV}_{\text{final year}} - \text{ESV}_{\text{initial year}}}{\text{ESV}_{\text{initial year}}}\right) \times 100 \tag{3.6}$$

where ESV = total estimated ecosystem service value. Positive values suggest an increase whereas negative values imply a decrease in amount. Moreover, we also estimated values of services provided by individual ecosystem functions within the study landscape using equation 3.7.

$$\text{ESV}_{\text{f}} = \sum (\text{A}_{\text{k}} \text{ x } \text{VC}_{\text{f k}}) \tag{3.7}$$

where ESV_f = calculated ecosystem service value of function ‘f’, A_k = the area (ha) and VC_{fk} = value coefficient of function ‘f’ (US$ ha^{-1} $year^{-1}$) for LULC type ‘k’. The contributions of individual ecosystem functions to the overall value of ecosystem services per year were ranked based on an estimated value of ecosystem functions for each reference year and summarized in a table.

Considering uncertainties exist in the value coefficients and since the biomes used as proxies for LULC types are not perfect matches in every case, sensitivity analyses were conducted to determine the percentage change in ESVs for a given percentage change in the value coefficient (Li et al., 2010). Accordingly, the ecosystem modified value coefficients for natural forest, plantation forest, croplands, grasslands, tree patches, woodlands and water bodies were each adjusted by 50% and the corresponding coefficient of sensitivity (CS) was calculated using equation 3.8 as in Kreuter et al. (2001), which is similar to the standard concept of elasticity in economics.

$$\text{CS} = \frac{(\text{ESV}_{j} - \text{ESV}_{i})/\text{ESV}_{i}}{(\text{VC}_{jk} - \text{VC}_{ik})/\text{VC}_{ik}} \tag{3.8}$$

where ESV_i and ESV_j = initial and adjusted total estimated ecosystem service values, respectively, and VC_{ik} and VC_{jk} = initial and adjusted value coefficients (US$ ha^{-1} $year^{-1}$) for LULC type ‘k’. If CS is greater than one, then, the estimated ecosystem value is considered elastic relative to that coefficient, but if CS is less than one, then, the estimated ecosystem value is considered to be inelastic, and the result will be reliable even if the value coefficient has relatively low accuracy. The greater the proportional change in the ecosystem service value relative to the proportional change in the valuation coefficient, the more critical is the use of an accurate ecosystem value coefficient (Kreuter et al., 2001; Li et al., 2010; Liu et al., 2012).

During the sensitivity studies, the value coefficients were also adjusted by consumer and producer price indices for analyzing impact of variations of value coefficients over time using equation 3.9. The obtained average changes in consumer and producer price indices was 4.9% (applied for non-market based services) and 5% (applied for market based services), respectively. These values were used to either discount the value coefficients which refer to 1994, if the period under consideration was before 1994, or compound the value coefficients, if the considered period was after 1994. The latter studies allow for a direct comparison of value coefficients and changes with the according gross domestic products for the periods investigated.

$$VC_{yk} = \sum (VC_{fk} \text{ x } (1 + CPI)^t \tag{3.9}$$

where VC_{yk} = time development value coefficient of a reference year for LULC type ‘k’, VC_{fk} = value coefficient of function ‘f’ (US\$ ha^{-1} year^{-1}) for LULC type ‘k’, CPI = average change in consumer price index in percentage applied for non-market based services while for market based services average producer price index (PPI) in percentage was used, and t = number of years between 1994 and the year under consideration.

3.3.4 Scenario modelling of LULC changes

A GIS-based simulation modelling approach was employed to compose datasets mainly generated from own research (Kindu et al., 2013, 2015, 2016, see Publications I, II and III), other sources (Table 3.1) and assumptions as followed by previous studies (Messina and Walsh, 2001; Grinblat et al., 2015) under three scenarios. A detailed description of the approach is given in the following section.

3.3.4.1 Drivers and ESVs of future LULC changes

Six major drivers that triggered changes of LULC for the past four decades in the study landscape and influenced the spatial distribution of LULC types were considered, namely population density, distance to road, distance to market, slope, rainfall and altitude (Kindu et al., 2015, see Publication II in the Appendix). A conservative value coefficient of the target LULC types modified by Kindu et al. (2016, see Publication III in the Appendix) was adopted to estimate the ecosystem service values and their changes in response to the future LULC dynamics of the studied landscape.

3.3.4.2 LULC Scenarios and future demands

Based on existing LULC-related policies in Ethiopia, local demographic information and historical LULC dynamics of the studied landscape, three future scenarios have been defined to predict LULC demand for 2050, namely *Business as Usual (BAU), Forest Conservation and Water Protection (FCWP)* and *Sustainable Intensification (SI)* scenarios (Table 3.5). The BAU scenario was designed mainly based on assumption of a continuation of LULC conversion rates of the past 40 years in the studied landscape (Kindu et al., 2013, see Publication I in the Appendix). Thus, prior to the demand calculations for 2050 in BAU scenario, the rate of changes between 1973 and 2012 for each LULC types were determined using equation 3.10 (Puyravaud, 2003):

$$r = \left(\frac{1}{(t_2 - t_1)}\right) \text{ x } \ln\left(\frac{A_2}{A_1}\right) \tag{3.10}$$

where r = rate of changes for each LULC type, and A_1 and A_2 = the extent of each LULC type at time t_1 and t_2, respectively.

Table 3.5: Summary of land use/land cover types and assumptions for demand calculations under the three scenarios in 2050.

	LULC types	Scenario 1- BAU	Scenario 2 - FCWP	Scenario 3 – SI
Assumptions	Bare lands, croplands, grasslands, natural forests, plantation forests, settlements, tree patches, water body and woodlands	Continuation of historical LULC changes	Strict implementation of spatial policies: no change allowed with in natural forests, plantation forests, woodlands and water bodies.	Strict implementation of spatial policies Change bare lands in to forests Better family planning Use improved seeds and fertilizers Restrict croplands in steep and very steep slopes

BAU = Business as usual; FCWP = Forest Conservation and Water Protection; SI = Sustainable Intensification

Afterwards, the trends of each LULC type during the last four decades were extrapolated using equation 3.11.

$$A_n = A_0 e^{rt} \tag{3.11}$$

where A_n = area estimate for each LULC type in year n, A_0 = area of base year, t = time period that is the difference between year n and year 0, and r = average annual rate of change.

The FCWP scenario was designed for only strict implementation of spatial policies about forest conservation (FDRE, 2007) and water body protection (FDRE, 2000) in the country. The forest policy has been approved in 2007 with a general objective of enhancing the economy of the country through appropriately conserving and developing forest resources. Similarly, the water policy, approved in 2000, has a general objective of ensuring water resources of the country are protected and utilized for the highest social and economic benefits of the peoples of Ethiopia. The Munessa-Shashemene forest of the study area is one of the forests designated as the National Forest Priority Areas (NFPAs) to stop further deforestation and forest degradation (Teketay et al., 2010). The water body (Lake Langano) is located in the lower part of the study landscape. The goal of FCWP scenario was to assess the effect of strictly implementing those existing spatial policies into future LULC patterns and changes in the study area without considering other issues. Thus, this scenario assumed that the area of forests and water body will be excluded, leading to competition of the remaining LULC types to obtain the demands for LULC types in 2050.

The SI scenario is about sustainably producing more outputs with more efficient use of inputs while reducing environmental damage and building the flow of environmental services (Pretty et al., 2011). It is also about conserving and restoring natural resources. Under this scenario, it was assumed that better family planning of the projected birth rate of 3% and death rate of 0.85% (Garedew et al., 2012), crop productivity is likely to double with proper use of modern farm inputs (e.g. fertilizers and improved seeds), reducing average number of livestock for the households by half for improved grazing lands, restricting croplands on steep and very steep slopes (Kindu et al., 2013, see Publication I in the Appendix) through proper implementation of existing spatial policies (FDRE, 2000; FDRE, 2007), and rehabilitation of bare lands into forests to obtain the demands for LULC types in 2050. Projected population of 2050 with better family planning in combination with the current data on per capita areas of settlements and croplands were used to calculate the total croplands and settlements demand for this scenario.

3.3.4.3 Data analyses of scenario modelling

The spatial datasets for scenario modelling were prepared as described previously (see section 3.3.2.2) and shown in Figure 3.4 to make them suitable for statistical analyses. Afterwards, the datasets were analyzed in SPSS using binary logistic regression to identify the relations between a set of explanatory variables (drivers) and the actual LULC patterns as dependent variables. The logistic regression was used to indicate the probability of a certain grid cell to be devoted to a LULC type given a set of driving factors (Verburg et al., 2002). The goodness of fit (i.e. performance of the logistic regression model) was evaluated using the Relative Operating Characteristic (ROC) test statistics (Lesschen et al., 2005). ROC is a common measure for the goodness of fit in logistic regression to know how well the independent variables correctly predict the value of the dependent variable. A ROC value of above 0.5 indicates the spatial distribution of all of the LULC types (dependent variables) could well be explained by the selected drivers of LULC types (independent variables).

Based on the regression results, a probability map, also known as location suitability map, was produced for each LULC type. Conversion elasticity (ELAS) was specified by the user considering the actual LULC types. The conversion elasticity is an estimate of the conversion costs to account for the differences in changes between LULC types, e.g. an area with water body is not easily converted into croplands than those covered by forests. The value ranges between 0 (easy to convert) and 1 (difficult to convert). The value was defined based on user knowledge of the situation and adjusted during calibration of the model. Then, for each grid cell, the total probability was calculated for each of the LULC types based on the probability (suitability) maps from logistic model, elasticity of LULC change and the iteration variable using the equation 3.12 as in Verburg et al. (2002).

$$TPROP_{i,u} = P_{i,u} + ELAS_u + ITER_u \tag{3.12}$$

where TPROP = total probability of location i for LULC type u, $P_{i,u}$ = suitability of location i for LULC type u (based on logit model), $ELAS_u$ = the conversion elasticity for LULC u and $ITER_u$ = an iteration variable that is specific to the land use type and indicative for the relative competitive strength of the land use type. Finally, a conversion allowance in a matrix form was specified in which a number of conversions that are not realistic were excluded, e.g., protection areas or conversion of areas with water body into forests. At the end, allocation of changes in LULC was carried out in an iterative procedure that fulfil the demand of each scenario for the different LULC types based on the highest total probability for the considered grid cell (Figure 3.4).

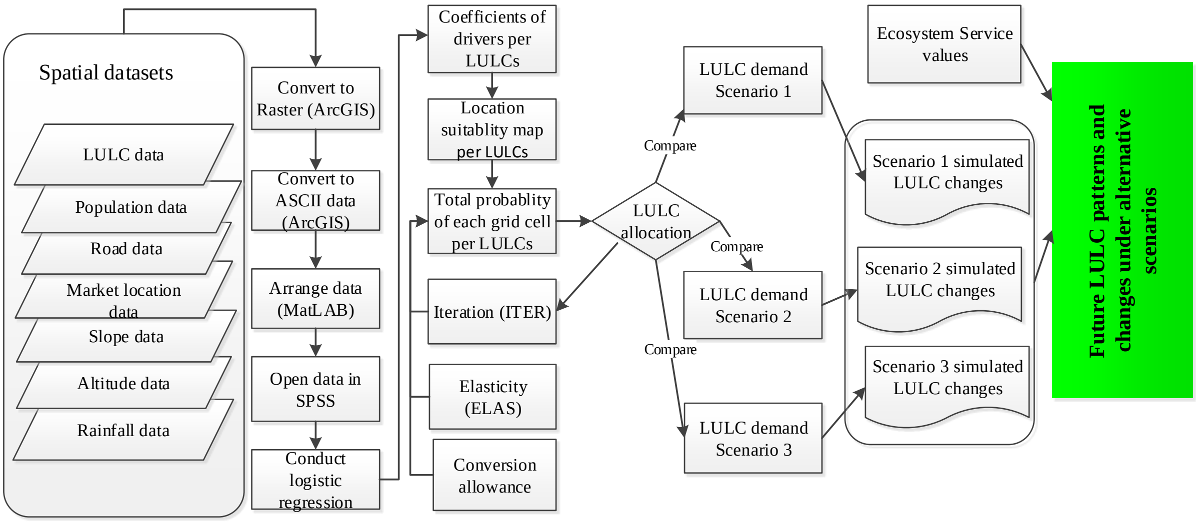

Figure 3.4: Methodological approach to investigate future LULC patterns and changes using scenario modelling.

Model validation was performed by generating a simulated map of 2012 LULC patterns in the studied landscape using the results of the logistic regression model, the transition matrix of the land use types and conversion elasticity. Then, the simulated LULC map was compared with the actual LULC map of 2012 derived from classification of satellite images (Kindu et al., 2013, see Publication I in the Appendix). Accordingly, overall accuracy, producer's and user's accuracies, and Kappa statistic were calculated from the error matrix. The Kappa statistic was used as one of the validation methods to evaluate the ability of the model to simulate the spatial patterns of LULC types (Pontius et al., 2001).

In order to estimate the ecosystem service values and their changes, the future LULC datasets of the considered scenarios were used as a proxy for the measurement and were prepared in the GIS environment. Accordingly, the total ESV of the landscape for each scenario, their percentage change and the values of services provided by individual ecosystem functions were estimated using equations 3.5, 3.6 and 3.7, respectively.

4 Results

4.1 States of land use/land cover (LULC) and their changes

4.1.1 States of LULC

The classification results revealed a total of nine LULC types extracted from satellite images of the year 1973, 1986, 2000, and 2012 (Figure 4.1, for area coverage, see Table 4.1). At the beginning of the study period (1973), grasslands were the dominant LULC type, making up 42.3% of the study landscape followed by natural forests (21%), croplands (13%), woodlands (11.4%) and the water body (9.6%). In 1986, similar order of extents was also accounted except for croplands and plantation forests (1%) that appeared during this particular reference year. The overall situation was changed in 2000. Croplands occupied the largest portion (39.5%), followed by grasslands (30.7%), natural forests (12.2%) and the water body (9.5%). The remaining portions were occupied by the other LULC types. In 2012, croplands continued to be the dominant LULC (48.5%) and the rest half of the study landscape was occupied by the other eight LULC types.

Table 4.1: Summaries of area of classified LULC types in the study area for the different reference years (adopted from Kindu et al., 2013).

LULC Type	1973		1986		2000		2012	
	Area (ha)	(%)	Area (ha)	%	Area (ha)	(%)	Area (ha)	%
Bare lands	343	0.3	1,462	1.4	1,598	1.5	1,765	1.7
Natural forests	21,726	21	16,065	15.5	12,680	12.2	9,588	9.2
Plantation forests	-	-	1,022	1	1,707	1.6	1,284	1.2
Croplands	13,498	13	31,178	30	40,997	39.5	50,317	48.5
Grasslands	43,830	42.3	38,606	37.2	31,853	30.7	25,139	24.2
Settlements	439	0.4	867	0.8	1,139	1.1	1,586	1.5
Tree patches	2,021	1.9	2,488	2.4	2,870	2.8	3,606	3.5
Woodlands	11,842	11.4	2,150	2.1	1,026	1	656	0.6
Water body	9,976	9.6	9,920	9.6	9,890	9.5	9,871	9.5

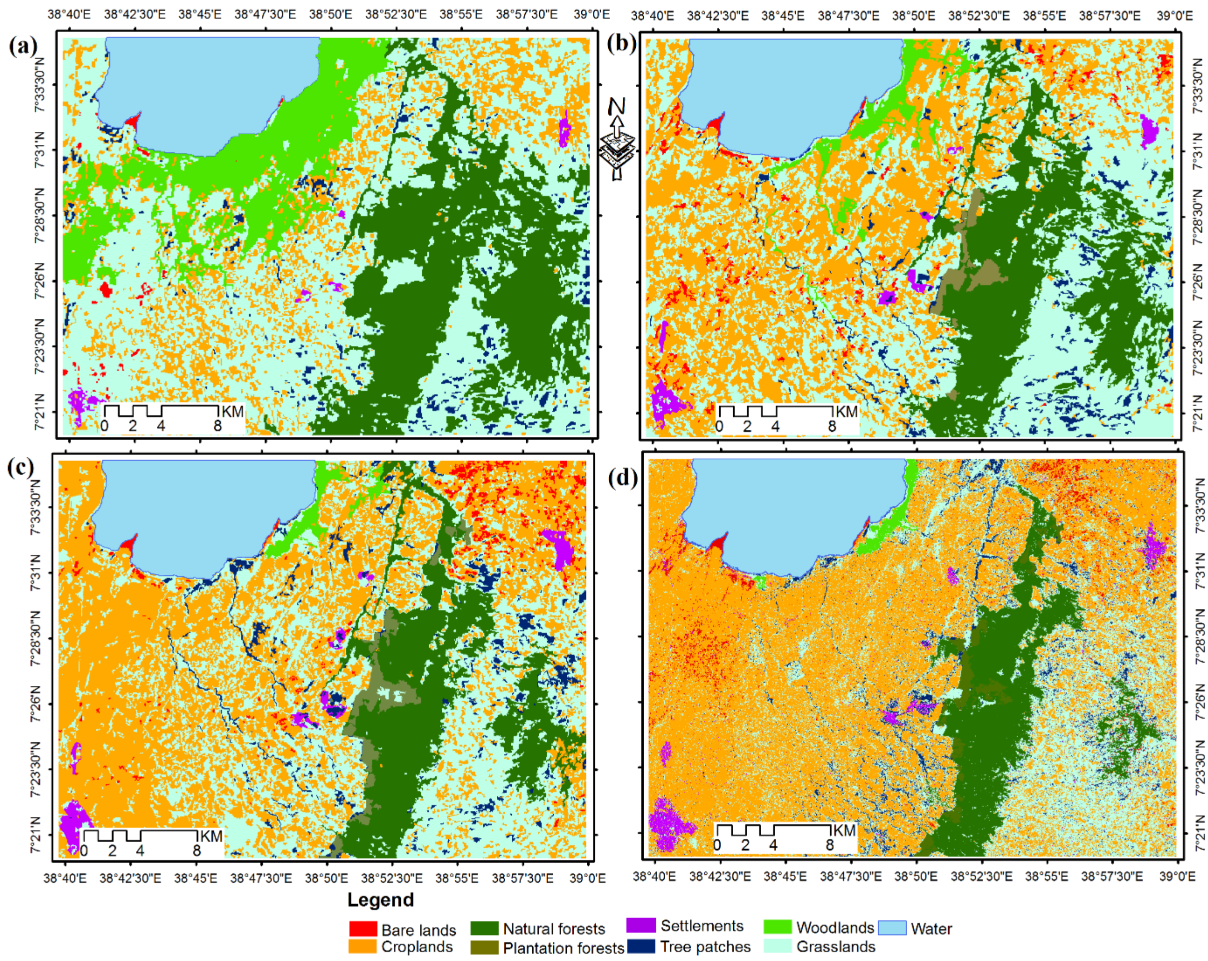

Figure 4.1: Land use/land cover (LULC) map of the study landscape **(a)** 1973, **(b)** 1986, **(c)** 2000 and **(d)** 2012 (adopted from Kindu et al., 2013).

4.1.2 LULC classification accuracy

The classification overall accuracies for the four reference years ranged from 85.7% to 93.2% with the Kappa statistic ranging from 0.82 to 0.92 (Table 4.2). The Kappa results show a high level of agreement for each of the four classified images.

Table 4.2: Summary of the classification accuracies for 1973, 1986, 2000 and 2012; where UA= user's accuracy and PA= producer's accuracy (adopted from Kindu et al., 2013).

LULC types	1973		1986		2000		2012	
	UA (%)	PA (%)	UA (%)	PA (%)	UA (%)	PA (%)	UA (%)	PA (%)
Bare lands	72.1	83.3	82.4	87.5	78.2	90.4	80.4	90
Natural forests	92.2	96.7	95.2	95.2	96.6	98.3	96.6	98.2
Plantation forests			96.3	94.5	98.2	96.5	98.2	96.4
Croplands	82.5	77.6	83.1	87.1	86.8	90.8	89.9	81.6
Grasslands	82.4	75.7	89.1	77	88.5	76.1	90.2	88.7
Settlements	75.0	90.7	85.5	92.2	87.5	89.1	89.5	94.4
Tree patches	91.4	82.8	93.4	90.5	93.5	93.5	98.4	96.9
Woodlands	90.3	84.8	94.1	98	93.8	97.8	97.9	97.9
Water body	95	98.3	98.1	100	98.1	100	100	100
Overall accuracy	85.7		90.7		91.9		93.2	
Kappa statistic	0.82		0.90		0.91		0.92	

4.1.3 Changes in LULC

The LULC change results revealed a considerable reduction of woodlands, natural forests and grasslands over the first (1973–1986), second (1986–2000) and third (2000–2012) study periods (Table 4.3). The woodlands converted by about 82, 52 and 36% of the cover that existed in 1973, 1986 and 2000 during the first, second and third periods, respectively. Similarly, the total area of natural forests and grasslands significantly decreased, whereas the water body showed a slight reduction during the study periods. On the other hand, croplands increased in all three periods by 131, 31.5 and 22.7%, respectively. Bare lands and settlements also increased in the three periods.

Table 4.3: Results of LULC changes from 1973 to 1986, 1986 to 2000 and 2000 to 2012 showing area changed for each class in hectare (ha) and percentage (%) (adopted from Kindu et al., 2013).

LULC Type	**Change in Land Use/Land Cover between Periods**					
	1973–1986		**1986–2000**		**2000–2012**	
	Area (ha)	**(%)**	**Area (ha)**	**(%)**	**Area (ha)**	**(%)**
Bare lands	1,119	326.2	136	9.3	167	10.5
Natural forests	−5,661	−26.1	−3,385	−21.1	−3,092	−24.4
Plantation forests	1,022	100	685	67	−423	−24.8
Croplands	17,680	131	9,819	31.5	9,320	22.7
Grassland	−5,224	−11.9	−6,753	−17.5	−6,714	−21.1
Settlements	428	97.5	272	31.4	447	39.2
Tree patches	467	23.1	382	15.4	736	25.6
Woodlands	−9,692	−81.8	−1,124	−52.3	−370	−36.1
Water body	−56	−0.6	−30	−0.3	−19	−0.2

The LULC changes matrix analysis in Table 4.4 shows that as a whole, about 61,848 ha (60%) of the land within the study landscape experienced changes in one or another way for the past four decades (1973–2012). The level of changes differed among the LULC types. For instance, out of 11,842 ha woodlands in 1973, only 651 ha (5.4%) remained unchanged during the study period, implying that about 95% of the woodlands were converted to other LULCs. Similarly, about 59% of the 1973 natural forests were converted to other LULC types. Of the 1,284 ha total cover of planation forests in 2012, about 946 ha (73.9%) was converted area, which used to be part of the natural forests in 1973. The original extents of grasslands were also reduced by 74.2% due to changes to other LULC types. Major opposite trends of changes were found for croplands, which gained an increase of 272% during the period analyzed (Table 4.4).

Table 4.4: Summary of LULC change matrix in ha from 1973 to 2012. [a] Row total sums areas for each LULC type of the initial study year (1973); [b] column total sums those areas of the year 2012. The bold diagonal values represent the area of each class that remains unchanged while the off diagonal values represent the change area (adopted from Kindu et al., 2013).

To \ From	BL	NF	PF	CL	GL	SL	TP	WL	WB	Total (2012) [b]
Bare lands (BL)	**79**	239	0	310.9	922	1.3	16.1	187.9	5	1,765
Natural forests (NF)	0	**8,922**	0	20.0	624	0	14.6	0	0	9588
Plantation forest (FP)	0	946	**0**	6.5	326	0	1.9	0	0	1,284
Croplands (CL)	227	3,193	0	**10,072**	27,871	47.8	937	7,828	14.1	50,317
Grasslands (GL)	21	7,226	0	2408	**11,756**	54	890	2,646	75	25,139
Settlements (SL)	14	45	0	335	805	**294**	13	70	8	1,586
Tree patches (TP)	0.9	1154	0	329	1,492	42	**146**	425	9.5	3,306
Woodlands (WL)	0	0	0	5.1	0.1	0	0	**651**	0.1	656
Water body (WB)	1.1	0	0	0.2	9.5	0	0.6	25	**9,834**	9,871
Total (1973) [a]	343	21,726		13,498	43,830	439	2,021	11,842	9,976	103,601

4.1.4 Distributions and changes of LULC along slopes and agro-ecological zones

LULC distributions and changes were remarkably different along the slope gradient (Figure 4.2). Almost all of the LULC types were found in the four slope gradients of the study landscape with different proportions. Croplands, grasslands and the water body were dominant LULC types in the gentle slope part. On moderate slopes, croplands, grasslands and the natural forests were found as the dominant land cover types. However, the natural forest and grasslands dominated the steep and very steep slope parts. Similar to distribution of LULC, percentages of changes were different along the slope gradient. The most significant overall trend was the shift of LULC proportions from gentle slopes (0%–5%) to very steep slopes (> 30%).

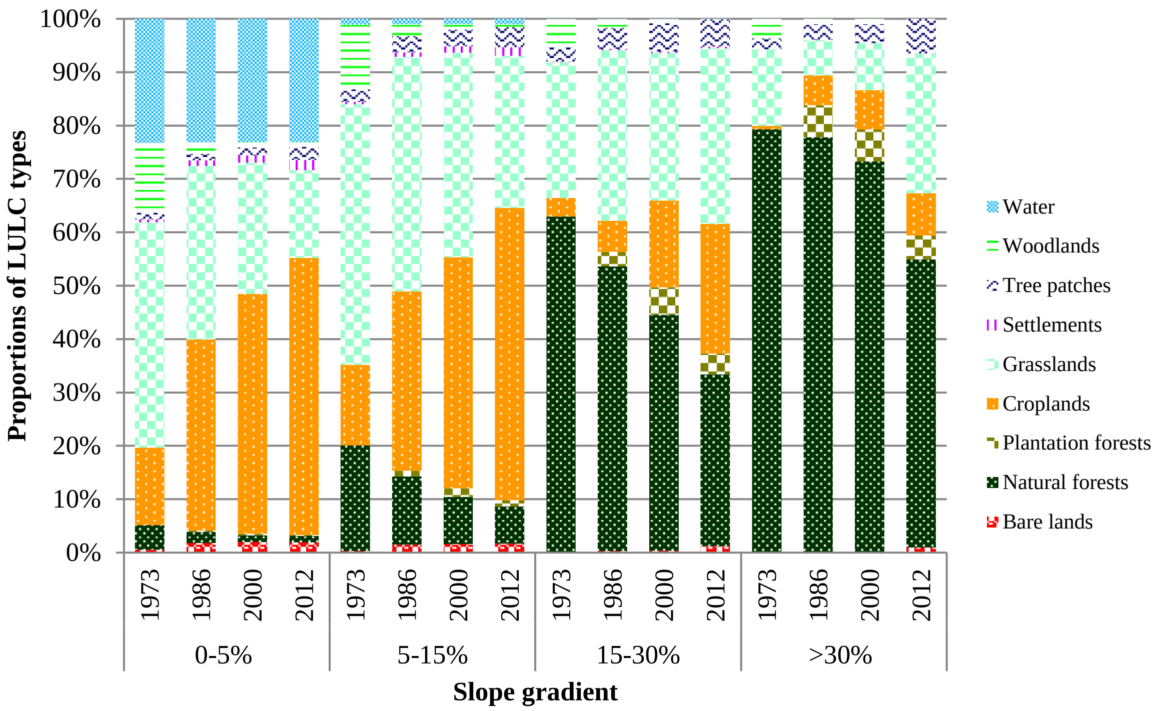

Figure 4.2: Proportion of LULC types in reference years along slope gradient: gentle slopes (0%–5%), moderate slopes (5%–15%), steep slopes (15%–30%), and very steep slopes (> 30%) (adopted from Kindu et al., 2013).

A distinct relationship between LULC and agro-ecological zones during the study period was also found (Figure 4.3). Among the three categories, diverse LULC types were observed in the sub-humid highland zone (locally known as "Weyna Dega", 1,500–2,300 m). All the classes of LULC types were found in this zone dominated by grasslands in 1973 and croplands in 2012. The numbers of LULC types were reduced in the humid highland ("Dega", 2,500–3,200 m) and cold highland ("Wurch", > 3,200 m) zones. The changes observed over the study period also differed notably along the agro-ecological zones associated with the elevation range. For instance, in the sub-humid highland zone, grassland showed the highest conversion from about 43% in 1973 to 17% in 2012 followed by woodlands from 17% to 1%, and natural forests from about 8% to 4%. The area covered by croplands increased from 15% to 56% during the same study period. In the cold highland zone, the area covered by the natural forest declined from about 78% to 23% while those of grasslands and croplands increased from about 21% to 52%, and from zero to 17%, respectively.

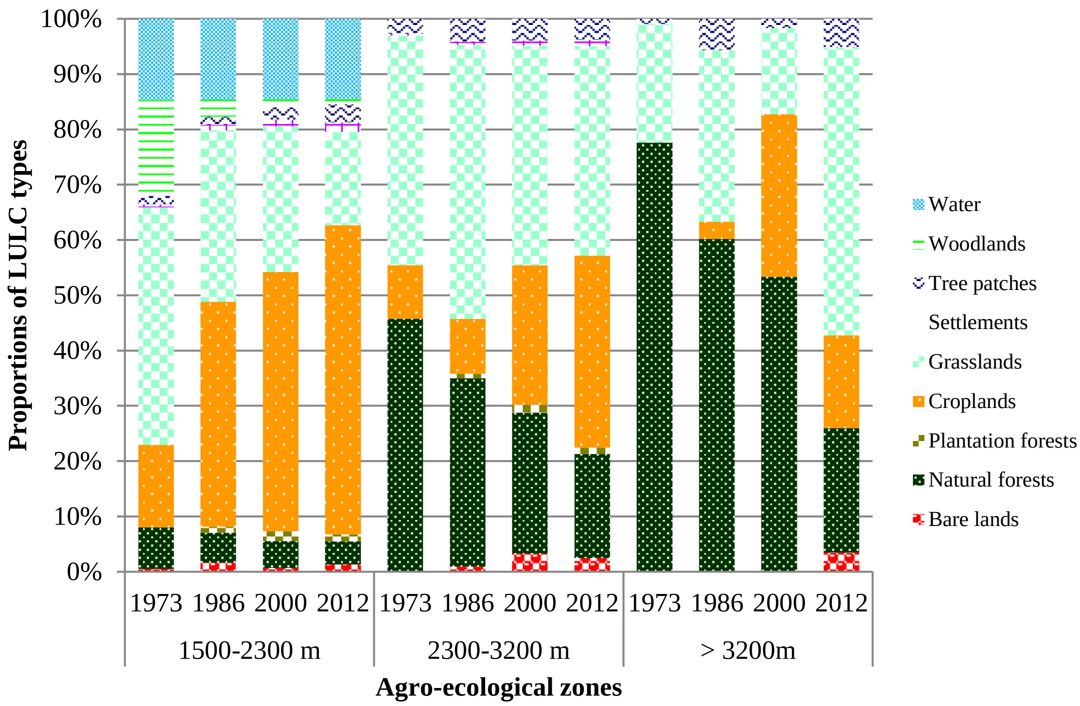

Figure 4.3: Proportion of LULC types in reference years along Agro-ecological zone: 1,500–2,300 m = sub-humid highlands (“Weyna Dega”), 2,300–3,200 m = humid highlands (“Dega”) and above 3,200 m = cold highlands (“Wurch”) (adopted from Kindu et al., 2013).

4.2 Understanding drivers of LULC changes

4.2.1 Main drivers of LULC changes

From a range of social, economic, environmental, policy/institutional and technological factors, a total of more than twelve factors were perceived by the respondents as being important drivers of LULC changes in the study landscape (Figure 4.4). However, there were variations about each of the factors to which the local people viewed as driver for the LULC changes. In particular, all of the respondents (N = 150) perceived population growth and cultivated land expansion as the main drivers of LULC changes, but not civil war and conflict. These respondents view much with population and croplands trends of the area, which increased by 395 and 272%, respectively, during the study period (Figure 4.5). During the focus group discussions (FGDs), elders attributed these to polygamous families. There are Muslims in the study landscape that still practice having more than two wives. They also pointed out that human influxes from the neighboring areas, mainly, during the beginning the

Socialist Government in Ethiopia, have intensified the pressure to the LULC change. In addition, the majority of the respondents confirmed house construction (settlement) (98.7%), fuelwood collection (97.3%), charcoal making (94.7%) and livestock ranching (94%) as also important causes for the observed LULC changes. It was understood during FGDs that cattle are allowed to graze on the remaining crop stalks on the croplands after harvest and on communal grazing lands. They also confirmed that there are farmers who are sending their cattle to the remaining forests, including woodlands because of the existing grazing land is below the carrying capacity of their livestock.

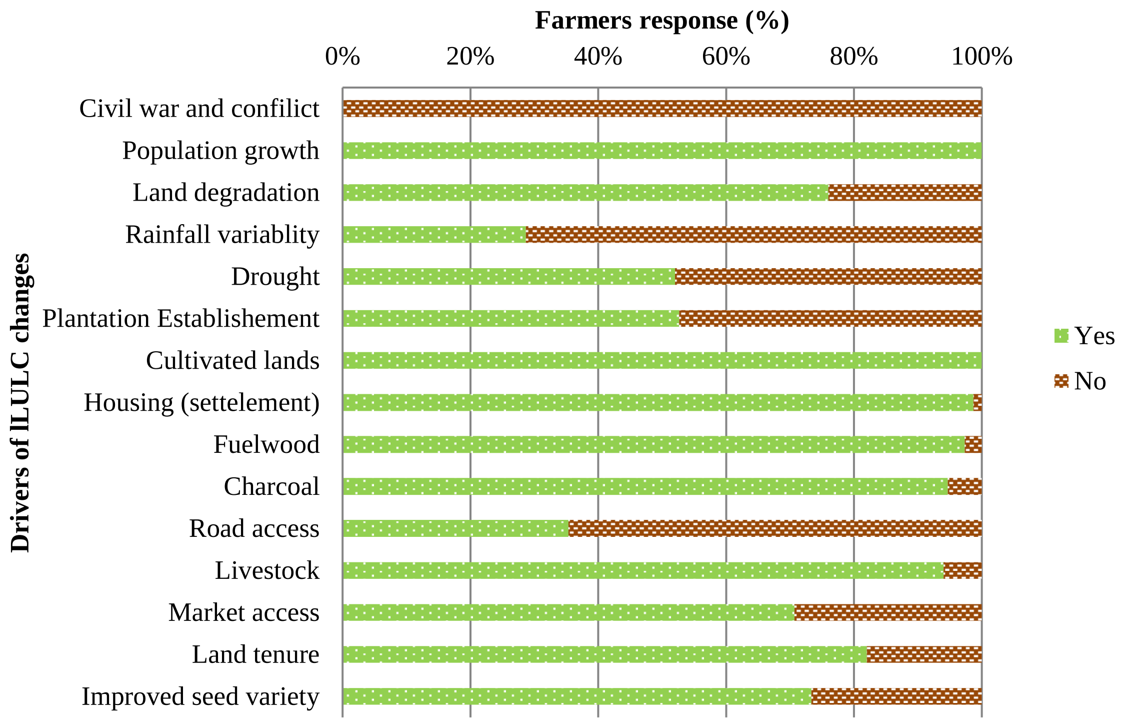

Figure 4.4: Drivers of LULC changes perceived by local people in Munessa-Shashemene study landscape (adopted from Kindu et al., 2015).

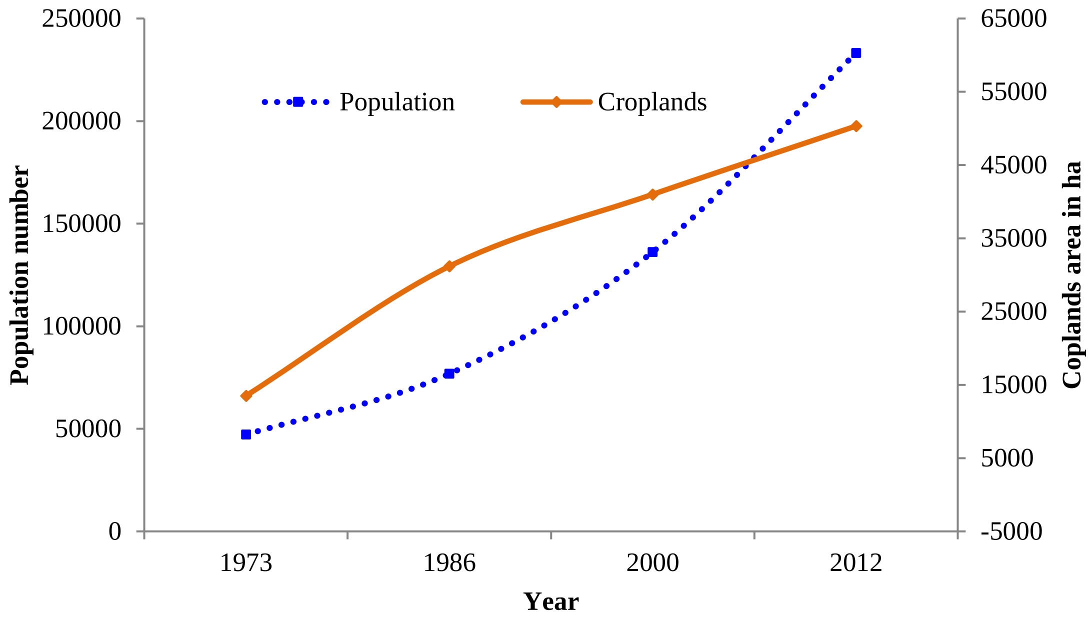

Figure 4.5: Population and croplands trends from 1973 to 2012 in the Munessa-Shashemene study landscape (adopted from Kindu et al., 2015).

Likewise, land tenure by 82%, land degradation by 76%, improved seed variety by 73.3% and market access by 70.7% of the respondents were considered as the other factors responsible for the prevalent LULC changes. During the FGDs, the participants described the changes in the government in 1974 (from Monarchy system to Socialist Government), which involved the 1975 land reform proclamation of ‘‘land to the tillers’’ triggered a change in land tenure from control by feudal landlords to peasant associations.

On the contrary, the respondents had less awareness on rainfall variability and road access as important drivers of LULC changes. In the first case, only 28.7% of the respondents believed that rainfall variability is a factor responsible for the changes, while in the second case, only 35.3% of the respondents considered road access as a driver of LULC changes. However, during FGDs discussion road access was mentioned as one of the influential drivers for changes. Similarly, the standardized growing season rainfall anomalies (SRA) in the lower part of the study landscape showed high variability in rainfall with a coefficient of variation of 30.3%. Of the 28 years of observation, 16 years (57%) had negative anomalies implying more dry years than wet years (Figure 4.6).

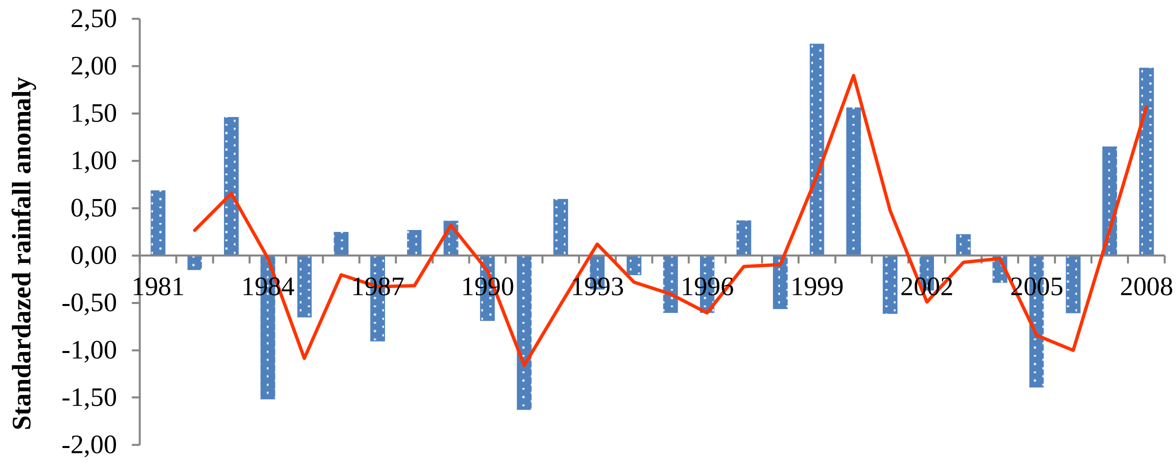

Figure 4.6: Standardized anomalies of growing season rainfall in Langano station at the lower of Munessa-Shashemene study landscape for the period 1981-2008 (red line 2 years moving average), CV = 30.3% (adopted from Kindu et al., 2015).

4.2.2 Ranked drivers of LULC changes and variations in perceptions among respondents

The results indicated that population growth was ranked first and as the most influential driver followed by expansion of cultivated lands, settlements, livestock ranching, cutting of woody plants for fuelwood and making charcoal as well as plantation establishment, which are listed in their descending order of priority (Table 4.5). The highest standard deviation of ranking was observed in plantation establishment. Road access and rainfall variability were viewed as the least influential drivers of changes with rankings of 13th and 14th, respectively.

Table 4.5: Respondents ranked drivers of LULC changes in the study landscape in order of influence (1 -14), with 1 being the most influential driver (adopted from Kindu et al., 2015).

LULC driver	N	Min.	Max.	Mean	Std. Dev	Rank
Population growth	150	1	3	1.05	0.24	1
Cultivated lands	150	1	5	2.14	0.48	2
Housing (settlement)	148	2	5	3.30	0.61	3
Livestock	141	3	11	4.84	1.20	4
Fuel wood	146	2	10	5.23	1.20	5
Charcoal	142	3	9	5.68	1.11	6
Plantation establishment	79	1	12	5.77	3.02	7
Land tenure	123	5	13	8.20	1.82	8
Market access	106	6	12	8.42	1.06	9
Land degradation	114	1	12	8.57	1.57	10
Drought	78	2	12	9.22	2.10	11
Improved seed variety	110	5	14	10.04	1.83	12
Road access	53	3	13	10.17	2.05	13
Rainfall variability	43	7	14	10.88	1.59	14

The perception towards drivers of changes significantly varied among respondents in different *Kebeles* concerning land degradation, rainfall variability, drought, planation establishment, road access, market access and improved seed variety as drivers of LULC changes but not others (Table 4.6). For instance, the majority of farmers in Degaga (96.7%) perceived plantation establishment as a major driver of LULC changes. Contrary to that, most farmers in Gorbi Arba *Kebele* did not consider plantation establishment as a key driver of change(s). Instead, the majority of these farmers perceived drought as a major driver of the observed changes. In a FGD, elders pointed out that they are dependent on selling charcoal and fuelwood as immediate source of income during decline or failure of crop production in drought years. However, most farmers in Komma Ocha *Kebele* did not consider drought as a major driver of changes (Table 4.6). The other drivers of LULC changes did not show any significant variation among surveyed respondents in the different *Kebeles*.

Table 4.6: Association/differences between farmer's perception responses towards drivers of LULC charges by Kebele-% of respondents (adopted from Kindu et al., 2015).

LULC change driver	Response by Kebele								
	Gorbi Arba (N= 30)		Dagaga (N = 30)		Gujicha (N= 50)		Kommo Ocha (N = 40)		X^2
	Yes	No	Yes	No	Yes	No	Yes	No	
Civil war and conflict	0	100	0	100	0	100	0	100	
Population growth	100	0	100	0	100	0	100	0	
Land degradation	100	0	96.7	3.3	62	38	60	40	***
Rainfall variability	56.7	43.3	40	60	20	80	10	90	***
Drought	96.7	3.3	40	60	38	62	45	55	***
Plantation Establishment	30	70	96.7	3.3	64	36	22.5	77.5	***
Cultivated lands	100	0	100	0	100	0	100	0	
Housing (settlement)	100	0	100	0	98	2	97.5	2.5	
Fuel wood	100	0	100	0	98	2	92.5	7.5	
Charcoal	100	0	96.7	3.3	94	6	90	10	
Road access	63.3	36.7	46.7	53	28	72	15	85	***
Livestock	100	0	93.3	6.7	100	0	82.5	17.5	
Market access	93.3	6.7	90	10	86	14	15	85	***
Land tenure	73.3	26.7	93.3	6.7	78	22	85	15	
Improved seed variety	90	10	60	40	94	6	45	55	***

*** Significant at $P < 0.001$, indicating that the location (Kebele) of farmers/households had a significant effect on the perception of farmers towards the drivers.

4.2.3 Household-level logistic regression of less perceived drivers of change

The household-level logistic regression results revealed that among the seven explanatory socio-economic determinants (variables) included in the analysis, educational level and age of the household head (respondent) had affected positively and significantly ($P = 0.001$) the low perception of farmers on rainfall variability as drivers of LULC changes (Table 4.7a). Among the farmers who did not perceive rainfall variability as a driver, 77.6 and 57% were illiterate and respondents with age lower than 50, respectively. Either literate or aged farmers were more likely to be aware of rainfall variability as a driver than the illiterate and younger farmers. The odds ratio also suggests that if a farmer is educated (literate), other factors held constant, the likelihood of awareness of the rainfall variability will be 58 times higher than an illiterate famer. Similarly, an increase in the age of a household head increases the likelihood of his/her perception of rainfall variability by a factor of 34.4.

Concerning the low perception of road access as a driver of change(s), educational level, age and occupation of the sample household head had significant effects (Table 4.7b). Educational level and age of the household heads had positive effects on their perception. Literate farmers or elders were more likely to be aware of road access as a driver than the others with odds ratio of 12.6 and 7.2 times, respectively. On the other hand, occupation had negative relationship with respondents' perception of road access as a driver of change(s). This implies that an increase in the engagement of a household head in farming activities decreases the likelihood of his/her perception of road access as a driver of change(s).

Table 4.7: Logistic regression results at household level for less perceived drivers: (a) rainfall variability and (b) road access in the study landscape (adopted from Kindu et al., 2015).

Independent variable	B	S.E.	Wald	df	Exp(B)
(a) Rainfall variability as a driver					
Sex	-0.82	0.63	1.69	1.00	0.44
Age	3.54**	1.05	11.27	1.00	34.44
Educational level	4.07**	1.07	14.41	1.00	58.40
Occupation	-0.91	0.84	1.17	1.00	0.40
Family size	-0.40	0.22	3.39	1.00	0.67
Land holding size	0.63	0.55	1.29	1.00	1.87
Income	0.00	0.00	0.34	1.00	1.00
Constant	-2.66*	1.27	4.38	1.00	0.07
R^2	0.48				
(a) Road access as a driver					
Sex	-0.70	0.62	1.28	1.00	0.50
Age	1.97**	0.60	10.86	1.00	7.19
Educational level	2.53**	0.61	17.03	1.00	12.56
Occupation	-4.61**	1.40	10.87	1.00	0.01
Family size	-0.38	0.20	3.52	1.00	0.68
Land holding size	0.07	0.52	0.02	1.00	1.08
Income	0.00	0.00	1.13	1.00	1.00
Constant	3.22*	1.45	4.91	1.00	25.03
R^2	0.45				

Number of observation = 150; R^2 = Negelkerke's R^2; B = Coefficient; S.E. = Standard error; Wald = Chi-Square; df = degrees of freedom; Exp(B) = odds ratio; statistically significant at *P <0.05 and **P < 0.001; Relative operation curve (ROC), i.e. correct predication = 80 (a) and = 80.7% (b).

4.2.4 Landscape-level logistic regression of drivers of change from 1973-2012

The landscape-level logistic regression results revealed important drivers of LULC changes with a significant effect (P = 0.001) from 1973-2012 (Table 4.8). Results with positive coefficient values of each of the drivers indicate increasing trends in the probability of LULC changes, and negative coefficient values suggest decreasing trends in the probability. From the results, a clear relation of the range of drivers can be observed with most of the signs of the coefficient values, which is in line with the hypothetical relationship. The further a pixel is from the road, the less likelihood of occurrence of LULC changes, and the closer a pixel is to a market, the greater the likelihood of occurrence of LULC changes. Also, the more the land occupied by croplands, the greater likelihood of occurrence of LULC changes, and the more bare land exists, the higher likelihood of occurrence of LULC changes. However, the results of slope coefficients were on the contrary, i.e. the greater the slope, the greater the likelihood of occurrence of LULC changes. The same was true for the agro-ecology variable; the higher the agro-ecological zone, greater likelihood of LULC change. These imply changes progressed to marginal areas, i.e. sleepy slopes and harsh agro-ecological zones since the other areas were already exhausted.

Table 4.8: Landscape-level logistic regression of drivers for LULC changes [time period 1973-2012, unit of observation: pixel (n = 287988)] (adopted from Kindu et al., 2015).

Variables	**B**	**S.E.**	**Wald**	**df**	**Exp(B)**
Gentle slope			3067.104	3	
Moderate slope	0.379	0.010	1485.318	1	1.460
Steep slope	0.775	0.015	2591.754	1	2.170
Very steep slope	0.555	0.025	506.584	1	1.743
Sub-humid highlands			251.721	2	
Humid highlands	-0.088	0.011	70.105	1	0.916
Cold highlands	0.532	0.044	148.262	1	1.702
Distance to road	-0.053	0.002	643.549	1	0.949
Distance to market	-0.015	0.001	318.709	1	0.985
Bare lands	4.050	0.069	3396.907	1	57.397
Croplands	2.439	0.012	40382.828	1	11.465
Grasslands	1.111	0.012	8313.859	1	3.037
Population density change	0.081	0.003	829.639	1	1.084
Constant	-1.007	0.020	2554.923	1	0.365
R^2	0.22				

Relative operation curve (ROC) = 73.2%, R^2 = Negelkerke's R^2; B = Coefficient; S.E. = Standard error; Wald = Chi-Square; df = degrees of freedom; Exp(B) = odds ratio. All variables are statistically significant at $P < 0.001$.

4.3 Estimating ecosystem service values (ESV) and their changes

4.3.1 States of estimated ESVs

The total ESVs of the whole study landscape were about US$ 130.5, 118.5, 114.8 and 111.1 million in 1973, 1986, 2000 and 2012, respectively (Table 4.9a and Figure 4.7a), when locally adapted coefficients were used. The amount of ESVs differed among LULC types of the entire study landscape with different reference years. When using the global coefficients developed earlier directly, the total ESVs of the whole landscape were about US$ 164.6, 135.8, 127.2 and 118.7 million in 1973, 1986, 2000 and 2012, respectively (Table 4.9b and Figure 4.7b). The amount of ESVs also differed among LULC types depending on different reference years.

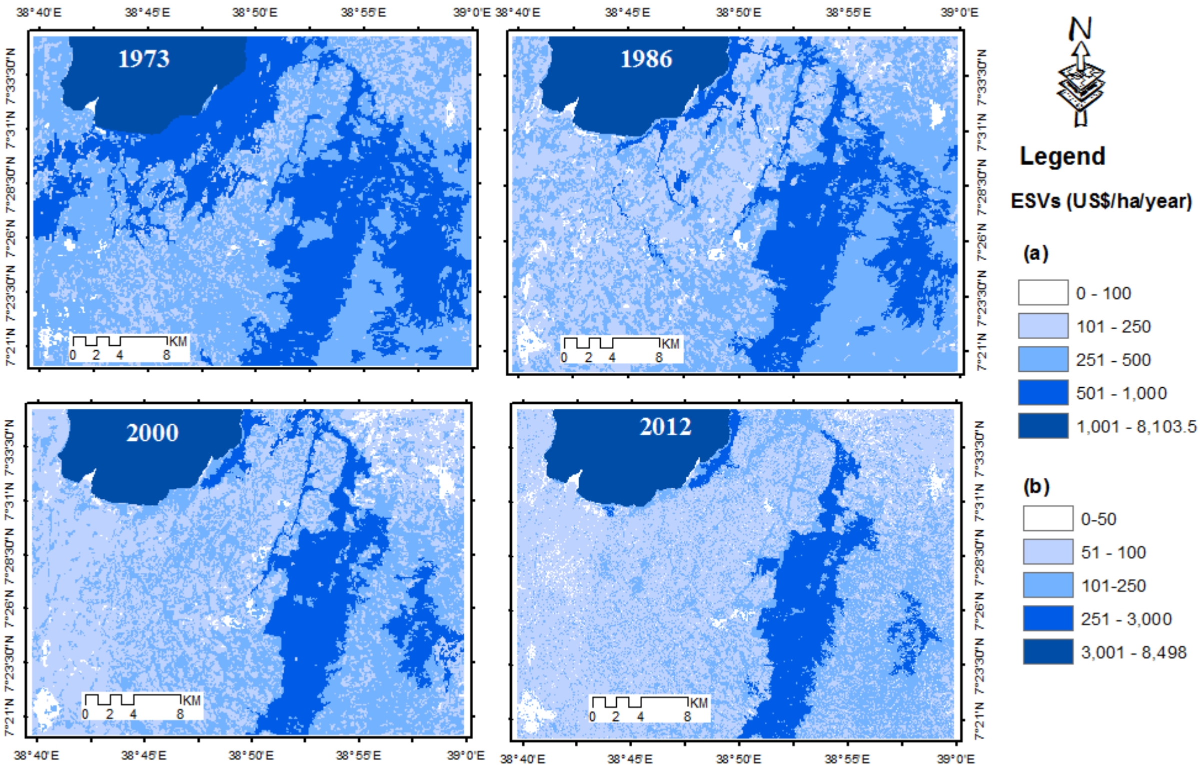

Figure 4.7: Spatial distribution of ecosystem service values ($/ha/year) in the study landscape in the reference years: (a) using modified conservative coefficients and (b) global coefficients from Costanza et al. (1997) (adopted from Kindu et al., 2016)

Table 4.9: Estimated ecosystem service values for each LULC type of the different reference years (in million US$ year^{-1}) in the study area using: (a) own modified conservative coefficients and (b) global coefficients from Costanza et al. (1997). Total ESV year^{-1} is given as a sum of each value by LULC type (adopted from Kindu et al., 2016).

LULC Type	1973		1986		2000		2012	
	ESV[a]	**%**	**ESV[a]**	**%**	**ESV[a]**	**%**	**ESV[a]**	**%**
a								
Bare lands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natural forests	21.4	16.4	15.9	13.4	12.5	10.9	9.5	8.5
Plantation forests	0.0	0.0	1.1	0.9	1.7	1.5	1.3	1.1
Croplands	3.1	2.3	7.0	5.9	9.3	8.1	11.4	10.2
Grasslands	12.9	9.9	11.3	9.6	9.3	8.1	7.4	6.6
Settlements	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tree patches	0.6	0.5	0.7	0.6	0.8	0.7	1.4	1.0
Woodlands	11.7	9.0	2.1	1.8	1.0	0.9	0.7	0.6
Water body	80.8	62.0	80.4	67.9	80.1	69.8	80	72.0
Total	130.5	100	118.5	100	114.8	100	111.1	100
b								
Bare lands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natural forests	43.6	26.5	32.3	23.8	25.5	20.0	19.3	16.2
Plantation forests	0.0	0.0	2.1	1.5	3.4	2.7	2.6	2.2
Croplands	1.2	0.8	2.9	2.1	3.8	3.0	4.6	3.9
Grasslands	10.7	6.5	9.4	6.9	7.8	6.1	6.1	5.2
Settlements	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tree patches	0.5	0.3	0.6	0.4	0.7	0.6	0.9	0.7
Woodlands	23.8	14.4	4.3	3.2	2.1	1.6	1.3	1.1
Water body	84.8	51.5	84.3	62.1	84.1	66.1	83.9	70.7
Total	164.6	100	135.8	100	127.2	100	118.7	100

[a]ESV = Ecosystem service value (million in 1994 US$ per year)

4.3.2 Changes of ESVs

The results of changes in ESVs revealed a significant reduction of the total values over the first (1973–1986), second (1986–2000), third (2000–2012) and whole (1973–2012) study periods (Table 4.10a). The total estimated loss of ESV was about US$ 19.3 million per year for the whole study period, which is about 14.8% of the value that existed in 1973. The change in ESVs of each LULC type also showed a reduction of the values from natural forest, woodlands, grasslands water body during the whole study period. On the contrary, the ESV of croplands increased during the same study period. With the global coefficients, although the amount differed, a significant reduction was also observed over the study periods, giving a total loss of US$ 45.9 million per year between 1973 and 2012 (Table 4.10b). This estimate is about 2.3 times higher than the loss estimated with the more conservative and better locally-adapted value coefficients. Consequently, the amount of changes of ESVs also differed among LULC types of the entire landscape in different reference years.

Table 4.10: Results of changes in ESVs from 1973 to 1986, 1986 to 2000, 2000 to 2012 and 1973 to 2012 time periods showing value changes [in million US$ for each LULC type and percentage (%) changes between periods] using: (a) own modified conservative coefficients and (b) global coefficients from Costanza et al. (1997) (adopted from Kindu et al., 2016).

LULC Type	Change in Ecosystem Service Values Between Study Periods							
	1973-1986		1986-2000		2000-2012		1973-2012	
	Million US$	(%)	Million US$	(%)	Million US$	(%)	Million US$	(%)
a								
Bare lands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natural forests	-5.6	-26.1	-3.3	-21.1	-3.1	-24.4	-12.0	-55.9
Plantation forests	1.1	100.0	0.7	67.0	-0.5	-24.8	1.3	100.0
Croplands	4.0	131.0	2.3	31.5	2.1	22.7	8.3	272.8
Grasslands	-1.5	-11.9	-2.	-17.5	-2	-21.1	-5.5	-42.6
Settlements	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tree patches	0.2	23.1	0.1	15.4	0.2	25.6	0.5	78.4
Woodlands	-9.6	-81.8	-1.2	-52.3	-0.4	-36.1	-11.0	-94.5
Water body	-0.5	-0.6	-0.2	-0.3	-0.2	-0.2	-0.9	-1.1
Total	-12.0	-9.2	-3.7	-3.1	-3.6	-3.2	-19.3	-14.8
b								
Bare lands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natural forests	-11.4	-26.1	-6.8	-21.1	-6.3	-24.4	-24.4	-55.9
Plantation forests	2.1	100	1.1	67.0	-0.9	-24.8	2.6	100
Croplands	1.6	131.0	0.9	31.5	0.9	22.7	3.4	272.8
Grasslands	-1.3	-11.9	-1.7	-17.5	-1.7	-21.1	-4.6	-42.6
Settlements	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tree patches	0.1	23.1	0.1	15.4	0.2	25.6	0.4	78.4
Woodlands	-19.5	-81.8	-2.3	-52.3	-0.7	-36.1	-22.5	-94.5
Water body	-0.5	-0.6	-0.3	-0.3	-0.2	-0.2	-0.9	-1.1
Total	-28.9	-17.5	-8.6	-6.3	-8.6	-6.7	-45.9	-27.9

4.3.3 Impact of LULC changes on ESVs

The influence of LULC changes on ESVs notably differed among the LULC types as observed in the contributions of the area and ESV for each LULC type over the study periods (Figure 4.8). In general, the area of natural forests, woodlands, grasslands and the water body consistently decreased with varied proportions over the study periods. Consequently, the total ecosystem services significantly decreased. In particular, the cover change in the forest ecosystem (natural forests and woodlands) greatly affected the changes of the total ecosystem service values in the study area during the whole study period. For instance, in total, the ESVs of natural forests and woodlands declined by about US$ 23.8 million, while the total ESVs in the study landscape decreased by about US$ 19.3 million (Table 4.9a). On the other hand, the area covered by croplands significantly increased from 13 to 48.5%. However, the corresponding ESV shows only a slight increase from about US$ 3.1 million in 1973 to 11.4 million in 2012 compared with the area expansion of croplands. Using the global coefficients, the overall trends of ESVs as a result of LULC changes were similar (Figure 4.8).

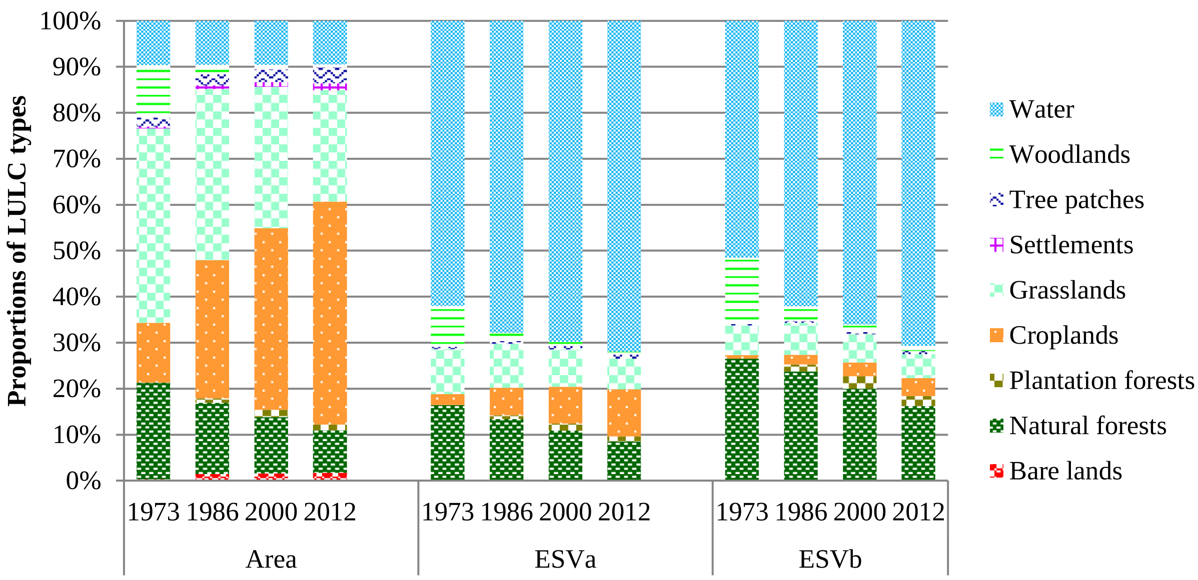

Figure 4.8: Area and ecosystem service value contribution of LULC types for different reference years in the study landscape [ESV_a= value obtained using modified conservative coefficients and ESV_b = value obtained by using the global coefficients from Costanza et al. (1997)] (adopted from Kindu et al., 2016).

4.3.4 Estimated services of individual ecosystem functions and their changes

The result showed that the first six individual ecosystem services, based on their contributions (from high to low) to the overall ESVs, were mainly from the service category of provisioning (water supply and food production) and regulating (water regulation, water treatment, erosion control and climate regulation) for the year 1973 (Table 4.11a). Their aggregated contribution represented about US$ 115.3 million, i.e. 88.4% of the total ESVs. This order of contribution changed over the different study periods. Additionally, contribution of individual ecosystem services declined throughout the study period, except for food production (Table 4.10a). Although the order of contributions of ecosystem services changed in the global coefficients, the value of major contributors declined during the study period (Table 4.11b).

Table 4.11: Estimated value of ecosystem functions (ESV_f in US$ million US$ year-1) under each service category for different reference years and their changes (1973 to 2012) in the study landscape (adopted from Kindu et al., 2016).

Ecosystem services	(a) ESV_f using Modified Conservative Coefficients					(b) ESV_f using Global Coefficients [a]				
	1973	1986	2000	2012	Overall Change	1973	1986	2000	2012	Overall Change
Provisioning Services										
Water supply	21.4	21.2	21.1	21.0	-0.4	21.4	21.2	21.1	21.0	-0.4
Food production	9.4	11.7	12.7	13.6	4.2	5.1	5.3	5.2	5.2	0.0
Raw material	1.7	1.0	0.8	0.6	-1.1	10.6	6.1	4.9	3.7	-6.9
Genetic resources	1.4	0.8	0.6	0.5	-0.9	1.4	0.8	0.6	0.5	-0.9
Regulating Services										
Water regulation	54.7	54.3	54.0	53.9	-0.8	54.7	54.3	54.0	53.9	-0.8
Water treatment	12.9	10.5	9.4	8.3	-4.5	13.5	11.8	10.9	10.1	-3.5
Erosion control	9.6	5.9	4.8	3.7	-5.9	9.6	5.9	4.8	3.7	-5.9
Climate regulation	7.5	4.3	3.4	2.6	-4.9	7.5	4.3	3.4	2.6	-4.9
Biological control	1.4	1.7	1.8	1.9	0.5	1.4	1.7	1.8	1.9	0.5
Gas regulation	0.8	0.6	0.5	0.4	-0.4	0.3	0.3	0.2	0.2	-0.1
Disturbance regulation	0.2	0.1	0.1	0.1	-0.1	0.2	0.1	0.1	0.1	-0.1
Supporting Services										
Nutrient cycling	6.2	3.5	2.8	2.1	-4.1	30.9	17.7	14.2	10.6	-20.3
Pollination	1.6	1.6	1.6	1.5	-0.1	1.3	1.5	1.4	1.4	0.1
Soil formation	0.4	0.2	0.2	0.1	-0.2	0.4	0.2	0.2	0.1	-0.2
Habitat/refugia	0.6	0.3	0.3	0.2	-0.4	0.1	0.2	0.2	0.2	0.1
Cultural Services										
Recreation	0.9	0.8	0.8	0.8	-0.1	6.1	4.5	4.1	3.6	-2.6
Cultural	0.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Sum	130.5	118.5	114.8	111.1	-19.3	164.6	135.8	127.2	118.7	-45.9

[a] adopted from Costanza et al. (1997)

4.3.5 Ecosystem service sensitivity analysis

The coefficient of sensitivity (CS) was less than one in all cases indicating that the total ecosystem service values estimated in the study landscape were relatively inelastic (low sensitive) with respect to the modified conservative ecosystem value coefficients (Table 4.12). Overall, the CS indicated that the ESV estimation was robust in spite of uncertainties on the modified value coefficients.

Table 4.12: Percentage change in estimated total ecosystem service values and coefficient sensitivity (CS) after a 50% adjustment of the modified conservative service valuation coefficients (VC) (adopted from Kindu et al., 2016).

	1973		**1986**		**2000**		**2012**	
Change of value coefficient	**%**	**CS**	**%**	**CS**	**%**	**CS**	**%**	**CS**
Natural forests VC ± 50%	± 8.22	± 0.16	± 6.69	± 0.13	± 5.45	± 0.11	± 4.26	± 0.09
Plantation forests VC ± 50%	± 0.00	± 0.00	± 0.43	± 0.01	± 0.73	± 0.01	± 0.57	± 0.01
Croplands VC ± 50%	± 1.17	± 0.02	± 2.97	± 0.06	± 4.03	± 0.08	± 5.11	± 0.10
Grasslands VC ± 50%	± 4.93	± 0.1	± 4.78	± 0.10	± 4.07	± 0.08	± 3.32	± 0.07
Tree patches VC ± 50%	± 0.23	± 0.00	± 0.31	± 0.01	± 0.37	±0.01	± 0.48	± 0.01
Woodlands VC ± 50%	± 4.48	± 0.09	± 0.90	± 0.02	± 0.44	± 0.01	± 0.29	± 0.01
Water body VC ± 50%	± 30.98	± 0.62	± 33.93	± 0.68	± 34.91	±0.70	± 35.98	± 0.72

The sensitivity of ESV changes estimated by employing common conservative coefficients (million in 1994 US$ per year) and time development coefficients are shown in Table 4.12. By considering time variable coefficients between the study periods, change in total ESVs during the four decades was about US$ 45.6 million, which is a bit more than double the loss of the value estimated using common coefficients. This additional loss is attributed to the effect of inflation and changes in producer prices. To gain insight for the magnitude of ESV changes in the four decades with time adjusted coefficients in comparison with GDP, the studied landscape GDP increased from about US$ 5.8 million year^{-1} in 1973 to 126.2 million year^{-1} in 2012 (Table 13). This implies that the total ESV loss was left with values of 2.1 times the GDP in 2012.

Table 4.13: Effect of using constant coefficients (in million in 1994 US$ year-1) versus time development coefficients between years to the overall estimation of ESV changes (adopted from Kindu et al., 2016).

Approaches	**Total ecosystem service values per reference year**				Overall changes
	1973	1986	2000	2012	
ESV: using constant coefficients with LULC dynamics	130.45	118.45	114.78	111.14	-19.3
ESV: using time development coefficients with LULC dynamics	47.69	80.72	153.04	263.51	
ESV: using time development coefficients with assuming no LULC dynamics	47.69	88.91	173.91	309.07	
Overall changes: using dynamics and time development coefficients					-45.55
GDP	5.8			126.2	

ESV = Ecosystem service value (in million US$ year^{-1}); time development coefficients were adjusted using consumer price index and producer price index; GDP = Gross Domestic Product of the study landscape (in million in 2012 US$ year^{-1}).

4.4 Scenario modelling of future LULC patterns and changes

4.4.1 Coefficients of logistic regression

The spatial distributions of the nine LULC types were well explained by the selected drivers of LULC types, as indicated by the relative operation curve (ROC) values that measure the goodness-of-fit of the logistic regression models (Table 4.14). From the result, the obtained ROC values for all the LULC types were above 50%. This implies the probability distribution generated using the considered drivers of LULC types were consistent with the actual distribution of the LULC types in the studied landscape.

Table 4.14: Beta (β) values for regression results of the spatial distributions of LULCs in the studied landscape.

Drivers	Land use/land cover types								
	Bare lands	**Crop lands**	**Grass lands**	**Natural forests**	**Plantation forests**	**Settle ments**	**Tree patches**	**Water**	**Wood lands**
Slope	-0.0231	-0.0379	-0.0044	0.0931	0.0251	-0.0472	0.0169	-0.1671	0.1610
Altitude	0.0003	-0.0021	0.0011	-0.0060	-0.0042	-0.0008	0.0002	-0.1149	-0.0671
Rainfall	-0.0005	0.0021	-0.0013	0.0239	0.0125	0.0010	0.0012	-0.0530	0.0257
Distance to road	-0.2517	-0.1089	-0.0228	0.3542	-0.1246	-0.3871	0.0521	0.5561	0.4526
Distance market	-0.0525	-0.1170	0.0254	0.0747	0.0270	-0.0592	0.0071	0.7037	-0.5407
Population density	-0.0012	-0.0010	0.0001	-0.0534	-0.0438	0.0019	0.0005	-0.1509	0.0016
Constant	-2.6525	4.8178	-4.8194	-12.5696	-5.4299	-2.1891	-5.2306	20.0617	9.1793
ROC values	0.6833	0.7443	0.6921	0.9449	0.8528	0.8473	0.6267	0.9998	0.8473

β represents the regression coefficients between LULC types and the drivers; ROC = relative operation curve; all drivers (variables) are significant at P <0.05.

4.4.2 Model validation

The overall accuracy for the simulated 2012 LULC map was 90.3% with the Kappa statistic of 0.889, which revealed the consistency between the simulated result and actual LULC situation (Table 4.15, Figure 4.9). The Kappa results show a high level of agreement for the simulated LULC map, indicating that the model is reliable for the studied landscape and can be used to simulate LULC patterns and changes under different scenarios. User's and producer's accuracies of individual classes of the simulated LULC map ranged from 72.9% (bare lands) to 92.6 (water body), and 78.5 (grasslands) to 94.3% (water body), respectively. This reveals that although more similarity is found between the simulated result and the actual LULC in 2012, it still exhibits a certain bias. The producer and user accuracies of the simulated map for bare lands and grasslands (producer's accuracy) were below 79%. On the other hand, the natural forests, plantation forests, tree patches, settlements, croplands, woodlands, and water body classes were relatively similar to the corresponding classes in the actual land use map for 2012.

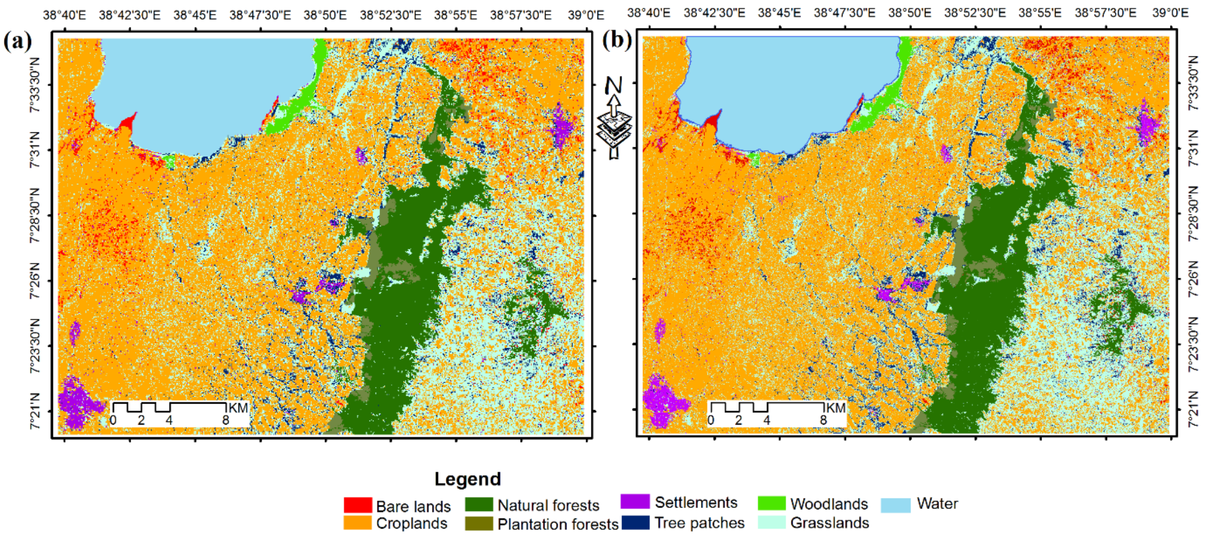

Figure 4.9: Actual **(a)** and simulated **(b)** land use/land cover map of the year 2012 in the study landscape.

Table 4.15: Summary of the classification accuracies for simulated LULC pattern in 2050; where UA= user's accuracy, PA= producer's accuracy, OA = Overall accuracy and KS = Kappa statistic.

	Land use/land cover type								
Accuracy	**Bare lands**	**Natural forests**	**Plantation forests**	**Crop lands**	**Grass lands**	**Settle ments**	**Tree patches**	**Wood lands**	**Water body**
UA (%)	72.9	90	91.5	83.1	81	86.4	88.9	88	92.6
PA (%)	79.6	88.5	91.5	79	78.5	89.5	90.3	84.6	94.3
OA (%)	90.3								
KS	0.899								

4.4.3 Simulation of land use/land cover patterns under different scenarios

The simulation results of LULC distributions for the considered scenarios illustrate different patterns in the studied landscape (Figure 4.10). In all of the scenarios, cropland was the dominant LULC type. Under the BAU scenario (Figure 4.10a), the trend of expansion of croplands is clearly along the direction of sloppy areas by replacing the remaining forest areas and grasslands. Comparing the simulated scenarios for 2050 with the actual LULC map of 2012, the croplands occurred primarily towards the eastern part of the studied landscape, making up 71.9% of the study landscape followed by the water body (9.4%), grasslands (6.3%)

and bare lands (3.7%) (Table 4.16). Tree patches, settlements, plantation forests, natural forests and woodlands shared small proportion of 2.8, 2.4, 2, 1.9 and 0.02%, respectively, of the entire landscape. Under the FCWP scenario, croplands also accounted for the largest part (66.3%); and the water body, natural forests, and grasslands accounted 9.5%, 9.2% and 5.4%, respectively, of the study landscape. Woodlands, tree patches, settlements, plantation forests, and bare lands occupied the smallest portion of the area. The results of the LULC pattern for the SI scenario were similar to those of FCWP scenarios, but the area proportions were better distributed for the different LULC types. Accordingly, croplands occupied (51.1%); and grasslands, natural forests, the water body and tree patches accounted for 16.2, 13, 9.5 and 4.1%, respectively, of the entire landscape. The remaining LULC types shared small portions of the study landscape.

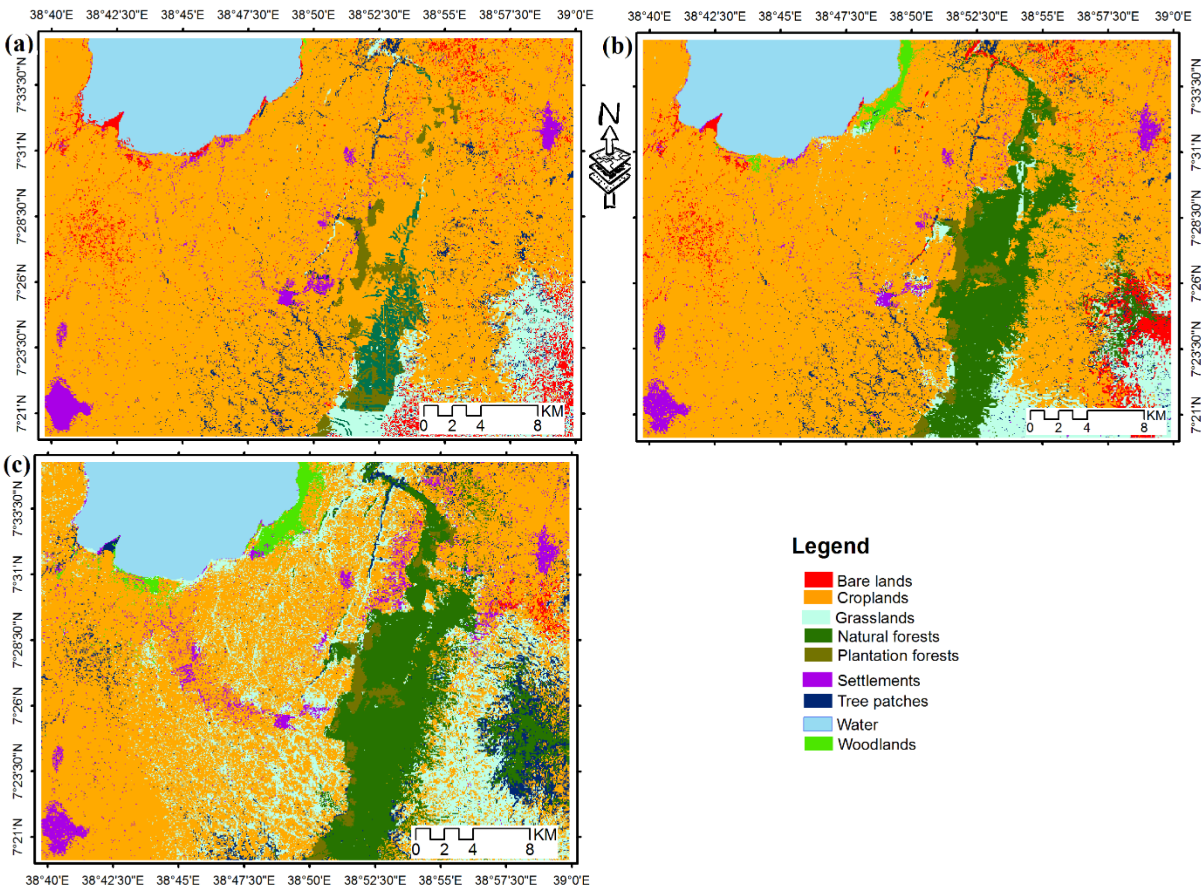

Figure 4.10: Simulated land use/land cover pattern in 2050 under different scenarios in the study landscape **(a)** Business as Usual (BAU), **(b)** Forest Conservation and Water Protection scenario (FCWP) and **(c)** Sustainable Intensification (SI) Scenarios.

Table 4.16: Summaries of areas of simulated land use/land cover types in 2050 under different scenarios in the studied landscape.

LULC types	Actual LULC 2012		Simulated LULC in 2050 under different scenarios					
			BAU		FCWP		SI	
	ha	%	ha	%	ha	%	ha	%
Bare lands	1765	1.7	3792.9	3.7	3301.1	3.2	343.0	0.3
Natural forests	9588	9.24	1928.7	1.9	9588	9.2	13549	13.1
Plantation forests	1284	1.24	2073.5	2.0	1284	1.2	1284	1.2
Croplands	50317	48.5	74205.8	71.6	68691.4	66.3	52982.2	51.1
Grasslands	25139	24.2	6529.6	6.3	5637.6	5.4	16823.7	16.2
Settlements	1586	1.53	2509.9	2.4	2167.0	2.1	3187.0	3.1
Tree patches	3606	3.47	2871.9	2.8	2479.5	2.4	4279.0	4.1
Woodlands	656	0.63	17.8	0.02	656	0.6	1356.3	1.3
Water body	9871	9.51	9745.5	9.4	9871	9.5	9871	9.5

BAU = Business as Usual; FCWP = Forest Conservation and Water Protection; and SI = Sustainable Intensification.

4.4.4 Changes in land use/land cover under different Scenarios

The change results revealed a considerable dynamics of woodlands, natural forests, grasslands, croplands and bare lands under BAU, FCWP and SI scenarios. Under BAU scenario, croplands and bare lands increased, while natural forests, woodlands and grasslands decreased (Figure 4.10 and Table 4.17). Changes in patterns of croplands predominantly occurred within major parts of the studied landscape distributed towards the remaining natural forests, woodlands and grasslands. The total woodlands converted under BAU scenario amounted to 638.2 ha, which is about 97% of the cover that existed in 2012. Similarly, natural forest cover and grasslands decreased by 79.9 and 74%, respectively. The areas of conversion were more on steep and very steep slopes, which are more susceptible to erosion. Thus, the dramatic change of forest resources would lead to ecological degradation in the studied landscape. On the contrary, croplands and bare lands increased by 47.5 and 114 %, respectively. In addition, the areas of settlements and plantation forests had increased.

Under the FCWP scenario, the study assumed LULC patterns and changes with only strict implementations of conservation and protection policies during the simulation years. Similar to

BAU scenario, the expansion trend for croplands occurred mainly in the east, excluding forest areas of the studied landscape. It increased to a value of 18,374 ha (36.5%), which is about 10% reduction compared to the BAU scenario. Consequently, grasslands decreased from 25139 ha in 2012 to 5,637.6 ha in 2050, which is about 77.6% of the cover that existed in 2012. The conversion trend of grasslands to croplands were also on steep and very steep slopes of the studied landscape, which indicated that Munessa-Shashemene still faces pressure of land degradation. However, the extent and distribution pattern of natural forests, plantation forests, woodlands and the water body were relatively similar to the conditions in 2012 of the studied landscape. The overall patterns and changes differed under SI scenario. Areas of croplands remained steady with gain of only about 5% of the cover that existed in 2012. In the same scenario, there were considerable gain in natural forests with a value of about 3,961 ha (41.3%) and woodlands with a value of 700.3 ha (106.8%).

Table 4.17: Simulated LULC changes from 2012 to 2050 showing the area changed for each classes in hectare (ha) and LULC changes in percentage (%).

LULC types	Simulated changes in LULC during 2012-2050 under different scenarios					
	BAU		FCWP		SI	
	ha	%	ha	%	ha	%
Bare lands	2027.9	114.9	1536.1	87	-1422.0	-80.6
Natural forests	-7659.3	-79.9	0.0	0.0	3961.4	41.3
Plantation forests	789.5	61.5	0.0	0.0	0.0	0.0
Croplands	23888.8	47.5	18374.4	36.5	2665.2	5.3
Grasslands	-18609.4	-74.0	-19501.4	-77.6	-8315.3	-33.1
Settlemts	923.9	58.3	581.0	36.6	1601.0	100.9
Tree patches	-734.1	-20.4	-1126.5	-31.2	673.0	18.7
Woodlands	-638.2	-97.3	0.0	0.0	700.3	106.8
Water body	-125.5	-1.3	0.0	0.0	0.0	0.0

4.4.5 Ecosystem service values and their changes under different Scenarios

The total ESVs of the whole study landscape were about US$ 102.4, 109.2 and 114.1 million under BAU, FCWP and SI scenarios, respectively (Table 4.18). The total estimated loss of changes of ESVs under FCWP scenario was about US$ 1.9 million, which is about 1.7% of the 2012 total ESVs. The total ESV was further decreased with an amount of about

US$ 8.7 million (7.8%) under BAU scenario. This estimate is about 4.6 times higher than the loss estimated under FCWP scenario. In both cases, the estimated losses during the four decades are less than 10% of the 2012 total ESVs, which is indicator of scarcity of ecosystems in the studied landscape. On the other hand, the SI scenario produced gains of the ESVs, which was increased with an amount of about US$ 3 million.

The estimated ESVs and their changes differed among LULC types of the entire study landscape under the considered scenarios. For instance, croplands accounted about US$ 16.7 million (16.3%) of the total ESVs, which is an increase of about US$ 5.4 million of the 2012 value under BAU scenario. The combined ESVs contribution of natural forests, woodlands and grasslands were about US$ 3.8 million (4%) of the total ESVs, while the losses were about US$ 7.6, 0.6 and 5.5 million of the 2012 value, respectively. Values of the water body also showed a slight reduction with an amount of about US$ 1 million and the ESV of croplands increased with an amount of about US$ 5.4 million (Table 4.18). Under FCWP scenario, estimated ESVs of grasslands decreased from about US$ 7.4 to 1.7 million, while the values of croplands increased from about US$ 11.3 to 15.5 million. This loss and gain of ESVs from the two ecosystems were improved under SI scenario. The situations in forest and woodland ecosystems have also positively affected the changes of ESVs. For instance, in total, the ESVs of natural forests and woodlands improved by about US$ 4.6 million, while the total ESVs in the study landscape improved by about US$ 3 million (Table 4.18).

Table 4.18: Estimated ecosystem service values for each LULC type of the considered scenarios of the study landscape (in million US$ year-1 and their changes in percentage (%) using conservative coefficients adopted from Kindu et al. (2016). Total ESV per year is given as a sum of each value by LULC type.

LULC Types	Actual -2012		BAU		FCWP		SI		Change in ESVs under considered scenarios					
									BAU		FCWP		SI	
	ESV	%	ESV	%	ESV	%	ESV	%	ESV	%	ESV	%	ESV	%
Bare lands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0		0.0	
Natural forests	9.5	8.5	1.9	1.9	9.5	8.7	13.4	11.7	-7.6		0.0		3.9	
Plantation forests	1.3	1.1	2.0	2.0	1.3	1.2	1.3	1.1	0.8		0.0		0.0	
Croplands	11.3	10.2	16.7	16.3	15.5	14.2	12.0	10.5	5.4		4.1		0.6	
Grasslands	7.4	6.6	1.9	1.9	1.7	1.5	4.9	4.3	-5.5		-5.7		-2.4	
Settlements	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0		0.0	
Tree patches	1.1	1.0	0.8	0.8	0.7	0.7	1.3	1.1	-0.2		-0.3		0.2	
Woodlands	0.7	0.6	0.02	0.02	0.7	0.6	1.3	1.2	-0.6		0.0		0.7	
Water body	80.0	72.0	79.0	77.1	80.0	73.2	80.0	70.1	-1.0		0.0		0.0	
Total	111.1	100	102.4	100	109.2	100	114.1	100	-8.7	-7.8	-1.9	-1.7	3.0	2.7

ESV = ecosystem service values (in million in 1994 US$ year^{-1}).

The water supply and food production of the service category of provisioning, and water regulation and water treatment of the service category of regulating were the top four service functions with the highest ESVs, contributing about US$ 94.9, 96 and 97 million under BAU, FCWP and SI scenarios, respectively (Table 4.19). The aggregated contribution of other thirteen ecosystem functions from each service category was about US$ 7.5 million, i.e. 7% of the total ESVs under BAU. When comparing the values with respect to service categories under BAU scenario, the results revealed the highest value for the group of regulating services (US$ 63 million), followed by provisioning services (US$ 36.6 million), supporting services (US$ 2.2 million) and cultural services (US$ 0.7 million). This order of contribution by service categories remained the same as the other scenarios. However, the amount of contribution by service categories declined under BAU and FCWP scenarios when compared with the 2012 values, except for provisioning services as a result of food production service function (Table 4.19). On the other hand, the contribution of individual ecosystem services showed an improvement of changes under SI in all service categories, mainly in the regulating service followed by supporting and provisioning services.

Table 4.19: Annual estimated value of ecosystem functions (ESV_f in million US$ year-1) under each service category for the considered scenarios in the study landscape using conservative coefficients (adopted from Kindu et al., 2016).

Ecosystem Services	ESV_{f-2012}		ESV_{f-BAU}		ESV_{f-FCWP}		ESV_{f-SI}	
	US$	%	US$	%	US$	%	US$	%
Provisioning Services								
Water supply	21.0	18.9	20.7	20.2	21.0	19.2	21.0	18.4
Food production	13.6	12.2	15.6	15.2	14.6	13.4	13.3	11.7
Raw material	0.6	0.5	0.2	0.2	0.6	0.5	0.8	0.7
Genetic resources	0.5	0.4	0.2	0.2	0.5	0.4	0.7	0.6
Regulating Services								
Water regulation	53.9	48.5	53.1	51.9	53.8	49.3	53.9	47.2
Water treatment	8.3	7.5	5.6	5.4	6.5	6.0	8.3	7.3
Erosion control	3.7	3.3	1.3	1.2	3.1	2.8	4.6	4.0
Climate regulation	2.6	2.3	0.9	0.9	2.6	2.4	3.6	3.2
Biological control	1.9	1.7	2.0	1.9	1.8	1.7	1.8	1.5
Gas regulation	0.4	0.3	0.1	0.1	0.2	0.2	0.4	0.3
Disturbance regulation	0.1	0.1	0.0	0.0	0.1	0.1	0.1	0.1
Supporting Services								
Nutrient cycling	2.1	1.9	0.7	0.7	2.1	1.9	3.0	2.6
Pollination	1.5	1.4	1.3	1.3	1.2	1.1	1.4	1.2
Soil formation	0.1	0.1	0.0	0.0	0.1	0.1	0.2	0.2
Habitat/refugia	0.2	0.2	0.1	0.1	0.2	0.2	0.3	0.2
Cultural Services								
Recreation	0.8	0.7	0.7	0.68	0.7	0.7	0.8	0.7
Cultural	0.02	0.0	0.01	0.0	0.02	0.0	0.03	0.0
Sum	111.1	100	102.4	100	109.2	100	114.1	100

By considering time variable coefficients between the simulation periods, the total loss of ESVs during the next four decades were about US$ 124.9 million under BAU and 26.9 million under FCWP scenarios while under the SI scenario, there will be total gain of ESVs worth about US$ 43.1 million (Table 4.20). They are more than 14 times the loss or gain of the values estimated using common coefficients, in which the higher changes of the values are attributed to inflation effects and changes in producer prices during updating value coefficients.

Table 4.20: Effect of using constant coefficients (in million in 1994 US$ year-1) versus time development coefficients of the simulated years to the overall estimation of ESV changes (method adopted from Kindu et al., 2016).

Approach	**Total ecosystem service values for actual and simulated years**				Overall changes: using constant coefficients with LULC dynamics		
	2012	BAU	FCWP	SI	BAU	FCWP	SI
ESV: using constant coefficients with LULC dynamics	111.14	102.43	109.24	114.10	-8.7	-1.9	3.0
ESV: using time development coefficients with LULC dynamics	263.51	1453.62	1551.65	1621.69			
ESV: using time development coefficients with assuming no LULC dynamics	263.51	1578.58	1578.58	1578.58			
Overall changes: using dynamics and time development coefficients	0.00	-124.96	-26.93	43.11			

ESV = Ecosystem service value (million US$ $year^{-1}$); time development coefficients were adjusted using consumer price index and producer price index.

5 Discussion

5.1 LULC and their changes

Monitoring of LULC changes provides critical inputs to evaluate complex causes and responses in order to project future trends better (Prenzel, 2004; Giri, 2005), and it is a prerequisite for making effective development plans (Gautam and Chennaiah, 1985; Carver et al 2012). Spatial distribution and changes of LULC over time were extracted in the highlands of Ethiopia, particularly focusing on the Munessa-Shashemene landscape by using satellite images of Landsat MSS (1973), TM (1986), ETM+ (2000) and RapidEye (2012), and by employing object-based image analysis. The overall accuracies obtained in this study were more than 91% except for Landsat MSS 1973 as a result of confusions associated with the lower spatial resolution and few bands available in Landsat MSS image (Lillesand and Kiefer, 2000; Dessie and Kleman, 2007). However, the overall accuracies obtained from all types of images were higher than the 85% minimum threshold, set by Anderson et al. (1976) and Thomlinson et al. (1999) for effective LULC change analysis. The overall accuracy of this study was better than other local studies with similar geographical settings, LULC types and satellite imageries. For instance, Dessie and Kleman (2007) achieved 87% accuracy using Landsat TM, Wondie et al. (2011) reported 88% accuracy using Landsat ETM+, and Shiferaw (2011) achieved 86.1% using Landsat ETM+. This study found above 91% over all accuracies using the same satellite imageries by using the advantage of applying object-based classification techniques, such as combining spectral- and spatial information for extracting target objects (Baatz and Schäpe, 2000).

The change analysis of 39 years with three time periods (1973–1986, 1986–2000 and 2000–2012) and a change matrix from 1973 to 2012 revealed the extent of changes that occurred in different LULC classes throughout the four decades. In general, the area of natural forests, woodlands, grasslands and water body consistently decreased with varied proportions over the study periods. The reductions of these LULC types were mainly attributed to the conversion of the areas to croplands. The conversion of grasslands, natural forests, and woodlands to croplands was quite intense, and common in the study landscape. Such trends are consistent with numerous studies in Ethiopia and elsewhere (e.g. Tekle and Hedlund, 2000; Zeleke and Hurni, 2001; Dessie and Kleman, 2007; Pare et al., 2008;

Shiferaw, 2011; Knoke et al., 2012). Contrary to these and the findings of this study, increase of forest cover was observed by Bewket (2002) in Chemoga watershed within the Blue Nile, which was attributed to community afforestation programs. The analysis also revealed that planation forests were created at the expense of the remaining natural forest, which belongs to tropical dry Afromontane forests (Fritzsche et al., 2007). Cyranoski (2007) reported similar conversion of natural forests to planation forests in Indonesia. Such conversion is detrimental to biodiversity conservation, unless deforestation is unavoidable and plantation forestry is a “lesser evil” (Brockerhoff et al., 2008). Changes were not limited to the forest resources, but also to the grasslands and water body of the study landscape. Similar trend was also reported by Assen (2011) in the East Hararghe Ethiopian Highlands.

LULC changes in relation to gradients of slopes show expansion of croplands from gentle and moderate slopes to steep and very steep slopes. A similar trend was also observed by Zeleke and Hurni (2001) in which cultivation expanded to marginal areas as steep as > 30% slope. Change analyses across agro-ecological zones also revealed that croplands largely occupied the sub-humid highland zone of the study landscape, which is the best zone for growing diverse agricultural crops (Zeleke and Hurni, 2001; Hurni, 1998). LULC changes from natural forests to croplands in cold-highland zone were not as severe as in the sub-humid highlands since the landscape has harsh environment for diverse crop production similar to the lower agro-ecological zone (Hurni, 1998). Although such environment receives less pressure on the natural forest for crop production, an increase of grassland cover was observed, which might be linked to conversion in search of additional land for grazing (Bishaw, 2009).

Such LULC change analyses entail to reconsider the massive program of natural resource conservation to reduce environmental degradation and poverty as well as increase agricultural productivity and food security over the last three decades undertaken by the government of Ethiopia and a consortium of donors (Kassie et al., 2010), and revert the ongoing situation by devising site-specific and more effective landscape management strategies.

5.2 Drivers of LULC changes

Identifying and understanding drivers of LULC changes is essential for modelling future dynamics or development of management strategies to ameliorate or prevent further decline of natural resources (Serneels and Lambin, 2001; Mottet et al., 2006). A range of drivers of LULC changes, which occurred for the last four decades, was identified in the study landscape. Although the level differed, local farmers perceived the majority of influential drivers of the changes. These drivers are within the broader categories of social, economic, environmental, policy/institutional and technological forces causing the changes at national, regional and global levels. They only varied in their levels and types of causes that trigger the changes. In the studied landscape, population growth, expansions of cultivated lands and settlements were the top significant drivers of change in LULC.

The study landscape population quadrupled in the four decades of the study periods. At the same time, to support the rising population, there was a need for extensive agricultural production with continuous crop cultivation. Earlier studies in other parts of the country and elsewhere also reported population pressure and expansion of agricultural crops as major drivers of LULC changes (e.g. Bewket 2002; Hamandawana et al., 2005; Hurni et al., 2005; Dessie and Kleman, 2007). The rapid increase of the population could be partially explained by the polygamy culture practiced in the study area. Similar trends were also observed in other African countries with polygamy culture (Hayase and Liaw, 1997). The situation seems more worrying as about 96% of the additional 2.4 billion of the 7.2 billion projected to inhabit the planet in the next 40 years will be from developing countries (UN DESA, 2013).

Other key drivers of LULC changes were expansion of settlements, fuelwood collection and charcoal making. The highest expansion of settlements was found during the first study period (1973–1986), where, especially, the 1985 national settlement policy for villagization program contributed (McCann, 1995). Similar trends as a result of settlement policy changes were also observed in other countries, including Ghana (Braimoh, 2004), Kenya (Campbell et al., 2005), Tanzania (Kassahun, 2011) and China (Long et al., 2008). Changes due to fuelwood and charcoal making are directly connected to the wide use of biomass-based energy source in the country (Tegene, 2002; Teketay et al., 2010; Asfaw and Demissie, 2012). Similarly, livestock ranching was the other contributory influential driver of LULC changes. Cattle are allowed to freely graze on communal grazing lands, the remaining crop stalks on croplands after harvest and in the remaining forests due to the low carrying capacity of the available existing grazing lands. This could result in grazing-induced degradation through hindering natural regeneration of the remaining forests (Teketay et al. 2010; Tesfaye et al. 2010). Land degradation and, hence, loss of land productivity and drought that caused severe food shortage, and the famine in 1973 and 1984 in Ethiopia were also responsible for the prevalent LULC changes (Taddese, 2001) similar to other African countries (Thiombiano and Tourino-Soto, 2007). Changes in land tenure policy also derived significant dynamics of LULC in the country in general (McCann, 1995) and to the studied landscape in particular. The land tenure policy of the country changed in 1975 due to the new land reform proclamation of "land to the tillers", which allowed peasants to have access to land for the first time. Consequently, landless people moved from one place to the other in search of land that resulted in additional pressure on the remaining forests (Dessie and Christiansson, 2008). Campbell et al. (2005) also reported a similar incidence of changes in LULC in Kenya due to changes in land tenure policy.

The study further revealed that among the main socio-economic determinants, educational level (literacy), age and occupation of household heads significantly affected their perception towards some of the drivers of LULC changes. In a case study site of eastern Ethiopia, Daba (2003) found age and literacy of households to be significant factors affecting their perception of land degradation as a problem, which is one of the drivers for the observed LULC changes. Similarly, Jadin et al. (2013) reported households that have better access to information were fully aware of drivers of forest cover dynamics in Vietnam. Apart from household level, landscape level analyses of potential factors revealed the contribution of each driver for LULC changes in the studied area. The result showed that location of LULC changes was determined by distance to major drivers of changes. For instance, it was observed that more changes in LULC occurred in areas closer to markets and roads. The proximity of the study landscape to one of the major roads connecting Addis Ababa to the southern part of the country and Kenya, and existence of the two district markets contributed to the observed LULC changes. This finding reconfirms a conclusion by previous studies that indicated the occurrence of more LULC changes in areas where there are improved road networks (Nagendra et al., 2003; Dessie and Kleman, 2007) and market access (Serneels and Lambin, 2001).

Among the LULC types, the forest ecosystems, which can be used to store carbon, to protect water balance and preserve biodiversity (Knoke and Hahn, 2013), and the grasslands were largely affected by the ongoing drivers of changes. Thus, the long-term effects of drivers of LULC dynamics in the studied landscape of the Ethiopian highlands could have negative environmental and socio-economic consequences unless intervention strategies are introduced.

5.3 Changes of ESVs

Studies on quantifying the changes of ESVs in response to LULC dynamics serve as an important tool to raise awareness (Liu et al., 2010) and contribute to developing knowledge on management of natural capital (Frélichová et al., 2014). The present study was the first of its kind in the highlands of Ethiopia that contributes to the evolving study of ecosystem service science by providing landscape level status of changes in ESVs from the region where datasets are limited. Currently, the use of proxy data is common in the estimation of ESVs and their changes (Chan et al., 2006; Yoshida et al., 2010; Costanza et al., 2014; Wang et al., 2014) in case of lack of baseline information (Maes et al., 2012). The approach of employing LULC datasets as a proxy of measurement facilitated the estimation process, as confirmed in previous studies (Wang et al., 2006; Li et al., 2010).

The global estimate of ESVs was proposed by Costanza et al. (1997). Afterwards, it has been applied by a number of researchers for similar studies (e.g. Kreuter et al., 2001; Zhao et al., 2004; Kubiszewski et al., 2013). However, the quality of such valuation depends on the accuracy of LULC classification (Konarska et al., 2002) and associated value coefficients (Costanza et al., 1997; Li et al., 2010). In this study, spatially representative datasets of classified LULC types generated from remotely sensed imagery with higher overall accuracies were utilized as a proxy of measurement (Kindu et al., 2013). However, the corresponding value coefficients were rough, criticized because of uncertainties and, usually, underestimate contributions of some LULC types.

The critiques arise due to the nature of value coefficients in monetary units used to estimate the total ESVs, as valuation involves ecosystem services both traded and not presently traded on markets (Daily et al., 2000; Zhao et al., 2004). The coefficient of valuation used in the present study, obtained through benefit transfer method, are based on earlier studies (e.g. Costanza et al., 1997) that derived the values either directly (market values) or indirectly (non-market values) by employing a wide variety of methods. Toman (1998) has pointed out in a critique of the Costanza et al. (1997) study that "… *there is little that can usefully be done with a serious underestimate of infinity …*" (p. 58). This means their estimate of the total value of all of the world's ecosystem services (US$ 33 trillion per year) does not reflect an incremental calculation as it is based on selected value per unit area multiplied by all the units in the biosphere. Thus, the aggregated figure leads to serious underestimate of infinity. He further argued that such calculation with aggregated estimates could suggest all aspects of nature in all places need the same level of safeguarding (Toman, 1998). In addition, existing valuation of services outside the market, such as travel cost methods, contingent valuations, replacement cost methods and hedonic pricing have serious pitfalls (Daily et al., 2000; Li et al., 2010). Each valuation methodology has its own strengths and limitations which restricts its applications. In the absence of site-specific valuation information, the benefit transfer method is an alternative to estimating ESVs (Kreuter et al., 2001; Troy and Wilson, 2006; Kubiszewski et al., 2013). Two ways of benefit transfer approaches are available: function transfer and value transfer (Czajkowski et al., 2017). Function transfer uses data available at the study site to predict value coefficients of a new site. Value transfer is an approach used to transfer the overall values from the original site to the new site. This method adapts the existing valuation information to new site and has also its own limitations and benefits, even if reported as performing the best as compared to the function transfer (Czajkowski et al., 2017). For example, it assumes that the study site is sufficiently similar to site from where the value coefficients is being transferred. But still, it can provide the means to compare the relative magnitude of changes in the provision of ecosystem services under different situations as, for example, in response to LULC changes. In this study, by acknowledging limitations that are as yet unresolved, attempt were made to carefully choose only value coefficients of tropical ecosystems which are similar to the study site characteristics.

The current available and widely used ecosystem service value coefficients also assume spatial homogeneity of services within ecosystems (i.e. LULC types). For instance, forests located either in steep slope or flat area received a similar estimation, although their ESV could greatly differ, for example concerning protection against erosion. Thus, the total ecosystem service value provided by forests would be altered depending on specific site characteristics or landscape complexity. The same is true for croplands and other LULC types found either in suitable or non-suitable area. That is, the used ecosystem value coefficients do, yet, not consider the environmental suitability of a particular land use/land cover (i.e. under the situation of LULC in the environmental land use conflict), which cause impacts to important ecosystem functions, including soil erosion, groundwater quality and habitat integrity (Pacheco et al., 2014; Valle Junior et al., 2014a, b; Valle Junior et al., 2015). Similarly, the value coefficients do not consider effect of scarcity, i.e. they are constant irrespective of the quantities of the services. Such factors are also important and should be taken into account in future development of ecosystem service value coefficients to improve the understandings of the distributed characteristic of ecosystem service values.

The present study also considers estimates of total ESVs, while marginal values would be relevant for decisions. The total ESVs are based on the value per unit area (value coefficients), that is, the sum of the producer and consumer surplus (meaning net of the cost of production) with a constant/static set of value weights, which are used by Costanza et al. (1997). The marginal economic values are related to an incremental increase in a set of ecosystem services over time and space without holding prices constant (Luisetti et al., 2014) and would be relevant for decisions. However, those exist only for market-based goods. A way to device marginal values for ecosystem services empirically would also be future direction of ESV estimates.

Despite such uncertainties, as also commented by different researchers (Limburg et al., 2002; Hein et al., 2006; Maes et al., 2012), the need of accurate coefficients is often less critical for time series than for cross-sectional analyses since value coefficients tend to affect estimates of directional change less than estimates of ecosystem values at specific points in time (Kreuter et al., 2001; Zhao et al., 2004; Tianhong et al., 2010). In addition, the classical sensitivity analysis indicated that the estimated ESVs for the study landscape were relatively inelastic, with respect to the value coefficients and relatively robust despite uncertainties in the value coefficients. This approach was proposed by Kreuter et al. (2001) and recently criticized by claiming erroneous interpretation (Aschonitis et al., 2016) though not yet abandoned (Zhang et al., 2017). The effect of updating value coefficients was also quite strong in this study, altering the estimated change in ESV from US$ -19.3 to -45.6 million per year. The coefficients were updated by means of consumer and producer price indices, which may only be considered a very rough method. Studies about development of willingness to pay with increasing GDP are rare, so that empirical evidence for the assumption of a positive correlation between consumer and producer prices in this study is yet not available. This study would suggest comparing GDP and ESV only when both are adjusted to the same year of comparison.

The results of this study revealed, in general, a decline of the total estimated ESVs in the study landscape throughout the four decades in both considered cases, i.e. when using the value coefficients by Costanza et al. (1997) and their modification. In particular, the ESVs of natural forests, woodlands, grasslands and the water body consistently decreased with varied proportions over the study years. The reductions of these ESVs were mainly attributed to the LULC dynamics. While the study showed that LULC dynamics in the study landscape resulted in a decline of ESVs over time delivered by the affected land, it also revealed that the loss depends on the interaction of changes in various LULC types. Specifically, the study suggested that the decline of ESVs were mainly linked to the huge conversion of forests, which is identified as the main provider of ecosystem services (Martínez et al., 2009). Earlier studies from elsewhere also mirror the findings of the present study that changes in LULC types by, particularly, altering the forest ecosystems have resulted in significant loss of ESVs. For instance, in Pingbian country, China, Li et al. (2007) reported a decrease of ESVs by 25.4% from 1974-2004 as a result of altered forest ecosystems. A recent study in West Africa also revealed a general decline of ESVs from 2000 to 2009 (Leh et al., 2013).

The study further revealed that the contribution of individual ecosystem functions declined throughout the study period. Significant changes have occurred in the values of specific ecosystem service functions such as erosion control, nutrient cycling, climate regulation and water treatment, which were among the highest contributors of the total ESVs from the category of regulating and supporting services. However, the value of food production services not drastically, but consistently increased during the study periods, which is mainly due to dramatic increase of croplands. Similarly, in Sanjiang Plain of northeast China, Wang et al. (2006) found changes between 1980 and 2000 favoring of a few ecosystem services (i.e. food production) at the expense of other services, such as disturbance regulation, cultural and recreation.

The estimates of ESV changes in response to LULC dynamics have brought valuable empirical evidences that can serve for several purposes. The study provided a means to compare and highlight the magnitude of changes of ecosystem services. However, caution should be taken about summing up ESVs for total estimate and not wrongly implement for decision making concerning protection of multiple ecosystem services. This may lead to compensatory effects between ESVs of target LULC types, where LULC extremely well performing in only one service will compensate strong underperformance of other services.

For example, the ESVs of water bodies are higher than forests. Huge conversion of forest resource under strict protection of water bodies in a landscape can favor some of the services (e.g. water provisioning) and disfavor others (e.g. erosion control). This could lead to environmental degradation of essential services in case of decision of land use planning made based on such total estimates without considering the performance of ESVs for each target LULC types. In this case, using normalized ecosystem services indicators for different LULC options has significant advantage for safeguarding multiple ecosystem services (Knoke et al., 2016). Despite the acknowledged limitations, the results can be used as the bases to raise awareness during policy formulations that can improve decisions involving various aspects of intervention strategies for sustainable use and management of land resources. It is also important to alert planners and decision makers to consider the need for revising the ongoing extensive agricultural practice at the expense of natural ecosystems.

5.4 Scenario modelling of future LULC patterns and changes

Understanding future LULC patterns and changes is a core for science-directed sustainable management of the resources. In this study, using LULC types and their associated driver datasets, the LULC patterns and dynamics for the next four decades was successfully simulated, mainly, by employing GIS technology in the studied landscape. Three kinds of scenarios, i.e. BAU, FCWP, and SI scenarios, were designed. The model is validated using the actual LULC map of 2012 and showed an overall accuracy of 90.3% with a Kappa statistic of 0.889, which is within the range of accuracies found in previous similar simulation studies (Echeverria et al., 2008; Kamusoko et al., 2011). It should be noted that the simulation approach of this study only addressed the future LULC patterns and changes based on the considered scenarios, which were constructed either empirically generated datasets or assumptions as followed by previous studies (e.g. Messina and Walsh, 2001; Soares-Filho et al., 2006; Teixeira et al., 2009; Grinblat et al., 2015). They are spatially explicit and also provide a representation of the entire landscape, i.e. modelling all major LULC types of the studied landscape. The approach, thus, overcomes the limitations of other simulation modelling studies that examine only a specific part of LULC types, i.e. urban growth modelling (Yang and Lo, 2003), crop rotation modelling (Wang et al., 2014) or grazing intensity modelling (Sabatier et al., 2010). However, it remains as a future scientific task, particularly comparison of the considered scenarios with mechanistic LULC modelling approach that have high potential to investigate the problem from a financial policies point of view, as suggested by Knoke et al. (2016).

The finding on the three scenarios revealed that croplands are likely to be continued as important and influential land use type in the future sustainable use and conservation of land resources in the Ethiopian highlands. They will tend to affect patterns of LULC types differently under the considered scenarios. Specifically, under the BAU scenario, future LULC changes are likely to be more pronounced around woodlands, followed by grasslands and natural forests as a result of, mainly, expansion of croplands. This unwanted development in the LULC system is explained in part by their spatial distribution. Woodlands occurred typically in lower elevations (below 1900 m) (Teketay, 2000) and are more of gentle to moderate slopes (Kindu et al., 2013), while natural forests tend to occur in the upper elevation areas, and often more of steep to very steep slopes, making them more unsuitable for crop cultivation (Tegene, 2002; Kindu et al., 2013). Thus, future expansions of croplands will tend to overtake first suitable areas and, then, unsuitable areas of the studied landscape, making woodlands, natural forests and grasslands the most vulnerable LULC types to spatial changes. This is consistent with recent findings on extensive agricultural impacts on the remaining forest resources in other tropical countries, e.g. in India (Tian et al., 2014), Zimbabwe (Baudron et al., 2011) and Tanzania (Estes et al., 2012).

Expansions of croplands also emerged as a major threat under the FCWP scenario. As a result of attempting only strict implementation of conservation and protection policies under this scenario, the pressure will be more to the grassland ecosystem that is also found on steep and very steep slopes (Kindu et al., 2013). This revealed implementation of policies targeting only specific component of LULC types (e.g. only forests or water body) will not be a solution for sustainable land resource use and conservation in the studied landscape. This may result in a failure similar to the previous efforts that have been made, where most of the remaining natural forests of the country were designated as National Forest Priority Areas (NFPAs) to stop further deforestation and forest degradation (Teketay et al., 2010). The natural forests of the studied landscape were part of the NFPAs though deforestation is still an ongoing process (Kindu et al., 2013). This problem is not limited to the landscapes of the Ethiopian highlands. Other landscapes in developing countries have also experienced similar situations (e.g. Petursson et al., 2013; Bax et al., 2016).

The SI scenario showed different LULC patterns and changes compared to the other two scenarios. Cropland remained steady in terms of expansion, owing to the assumption of intensification through improved farming and better family planning under this scenario. This scenario has also given a room for recovery of forest resources due to a side-by-side implementation of existing conservation and protection policies. As a result, the landscape could be more productive and multifunctional than the current one. In this regard, the finding of this particular scenario is also consistent with other studies that reported the role of diversified land use approach to balance food production and climate protection (Knoke et al., 2011; Schaldach et al., 2011; Knoke et al., 2013; Estrada-Carmona et al., 2014; Knoke et al., 2015).

The results also revealed a decline in total ESVs under the BAU and FCWP scenarios in response to future LULC patterns and changes. The estimated loss was quadrupled under BAU compared to FCWP scenario. These reductions of ESVs were mainly linked with the dramatic conversion of forests that are the main providers of ecosystem services (Martínez et al., 2009). Findings from elsewhere also showed that such changes are common in declining the corresponding ESVs (e.g. Li et al., 2007; Hu et al., 2008; Leh et al., 2013). On the other hand, the results showed improved total ESVs under SI scenario. This mirrored the findings of Wang et al. (2014) that reported an increase of total ESVs as a result of ecological restoration in Ningxia of China. Knoke et al. (2014) also found improved ecological and economic values as a result of afforestation and intense pasturing in abandoned farmlands of the tropical Andes in southern Ecuador.

The LULC patterns and changes under the considered scenarios in the studied landscape have different opportunities and constraints for resource use and conservation. As shown in the results under BAU scenario, the local people may continue with the usual way of crop cultivation for tapping the increasing cereal market opportunity of the country through exploiting the existing natural resources. For instance, expansions of croplands have been met at the expense of woodlands, natural forests and grasslands. Such kind of drastic change in LULC, however, will negatively alter the potential use of an area and may ultimately lead to loss of productivity (Tegene, 2002). This could, in turn, affect the local people by reducing the means of livelihoods of those who depend on land. Other constraint associated with the expansion of croplands at the expense of the forest resources, is the decline of biodiversity. The study landscape is known to have remnant dry Afromontane forests dominated by indigenous tree species among which some require high conservation priority (Tesfaye et al., 2010). The forest conversion over the modelled period of the BAU scenario in the study area could indirectly contribute to the loss of those high conservation priority species. Furthermore, expansion of croplands towards marginal lands on steep and very steep slopes will cause more runoff, will accelerate the soil erosion and may decrease water supplies in the absence of vegetation cover (Bewket, 2002). Such cultivation of steep and very steep slopes may result in serious degradation to the extent that the land could not sustain any crop production in the future as observed in other similar studies (Tegene, 2002; Lemenih and Teketay, 2004b). There is also a direct interaction between LULC changes and the hydrological process as well as the water ecology and quality of a given area (Baron et al., 1998; Arora, 2002; Beniston, 2003; Assen, 2011). Lake Alemaya, located in the lower plateau of the East Hararghe Ethiopian highland, had disappeared due to siltation as well as sediment accumulation, which is a result of deforestation (Alemayehu et al., 2007; Assen, 2011). Similarly, Lake Cheleleka, located in the southern central Rift Valley of Ethiopia, has dried out (Dessie and Johan Kleman, 2007). Although not drastically altered, the water body (Lake Langano), located in the lower part of the study landscape, was declining over the modelled period. Unless appropriate measures are taken urgently, the tragic loss similar to Lake Alemaya and Cheleleka could happen.

Attempting only strict implementation of conservation and protection policies under the FCWP scenario will decrease the croplands by 10% compared with the BAU scenario. This could, in turn, result in a deficit to feed the projected population of the studied landscape. As a consequence, the farmers living in the study landscape will be forced to migrate to the neighboring or other areas of the country. This is one of the common survival strategies of rural people in the events of food deficits due to various causes. For instance, droughts during the 1970s and early 1980s in northern Ethiopia have caused an influx of migrants to southwestern Ethiopia (Reid et al., 2000). Tsegaye et al. (2010) also reported drought events during 1973/1974 and 1984/1985 forced influx of migrants from the neighboring Tigray to the northern Rift Valley dry lands of Ethiopia.

Under SI scenario, degraded lands could be exploited as an opportunity to catalyze restoration of native flora (Lemenih and Teketay, 2004b). This is because the scenario has its largest effect in the restriction of expansion of croplands and rehabilitation of degraded lands. Furthermore, the results from SI scenario, which incorporated conservations and protection policies, better family planning (implying reduced birth rate), and improved crop production, suggested that such integrated approaches of resource management will increase the areas of forest resources in the future, particularly on steep and very steep slopes that are not suitable for crop cultivation. Such recovery of natural habitats will increase the ecosystem services (Martínez et al., 2009; Kindu et al., 2016). Existing opportunities of incentives for conservation efforts, such as payment for ecosystem services provided through REDD+ projects (Cerbu et al., 2013) or using the area as source of tourist attraction (Juutinen et al., 2014), may also be additional source of income to the local people that can be provided by this scenario.

The holistic approach of integrated landscape management for sustainable land resource use and conservation, represented by the SI scenario, will contribute to the ongoing efforts of improving agricultural production and biodiversity conservation, which are not, as yet, simultaneously implemented in an integrated way in various part of the country (Wale, 2008; Spielman et al., 2010; Shiferaw et al., 2014). The findings are also important as knowledge on the topic was previously restricted to historical LULC dynamics in the Ethiopian highlands (Bewket, 2002; Hurni et al., 2005; Kindu et al., 2013; Temesgen et al., 2013). It helped to unleash the opportunities and constraints provided by this holistic approach at landscape level to ensure economic development and ecosystem health in the studied landscape or other areas with similar settings.

6 Summary and conclusions

The present study developed a novel framework using the applications of geo-informatics (remote sensing and GIS) in a multi-disciplinary approach to better understand the process of natural resource dynamics for appropriate intervention strategies. The study investigated LULC dynamics of the past four decades (1973-2012), their drivers and changes of ESVs in response to the dynamics in Munessa-Shashemene landscape of the Ethiopian highlands. In addition, the possible future LULC patterns and changes covering the next four decades were simulated and examined using a spatially explicit GIS-based model under three scenarios, i.e. BAU, FCWP, and SI scenarios. The LULC changes were analyzed and examined by object-based classification of multi-temporal remotely sensed data (Landsat and RapidEye) from four reference years followed by post classification comparisons using recent advancements of geo-informatics technology. This is a vital technology for continuous monitoring of LULC changes at varied spatial and temporal scales, which are otherwise not possible to simply attempt through ground based inventory. The object-based image analysis approach used in this study is new for investigating Ethiopian landscapes having diverse features and ragged topography. It has the potential to be extended across other parts of the country for improved classification results. The generated datasets of this study also provided up-to-date LULC dynamics. The datasets will be inputs for spatial database that can be used as a model for monitoring of future changes, informed decision making during policy formulation or land use planning process or other similar studies in the Ethiopian highlands.

The quantitative change results revealed a continuous increase of croplands at the expense of decreasing natural forests, grasslands and woodlands. This indicates that the increase of other LULC types, mainly monoculture agricultural systems, in the study area is the result of deforestation and grassland conversion. Such kind of agricultural landscape may lead to environmental and ecological problems. The situations were clear in the analyses of the 39-year change matrix, which revealed that about 60% of the land had experienced changes in LULC. The results from the change analyses imply that horizontal expansion of croplands towards to the remaining natural forests that exist in very steep slopes, and harsh environment is no longer an option. Therefore, strategies to take urgent action against this will have to seek a sustainable solution that better address proper management of the resources.

The detected LULC changes that occurred in the studied landscape were due to, mainly, the interplay between more than twelve drivers related to social, economic, environmental, policy/institutional and technological factors. Specifically, population growth, expansion of cultivated lands and settlement areas (housing), livestock ranching, cutting of woody plants for fuelwood and charcoal making are the top six important drivers of LULC changes as viewed by the local people as well as from the detailed quantitative landscape level analyses. The findings further depicted that among the main socio-economic determinants at household level, educational level (literacy), age and occupation of household heads significantly affected their perception towards some of the drivers of LULC changes. Consequently, either aged or literate farmers were more likely to be aware of rainfall variability as a driver, whereas an increase in the household heads engaged in farming activities decreases the likelihood of their perception of road access as drivers. At landscape level, location of LULC changes were determined by distances to major drivers of changes (e.g. the further a pixel is from the road, the less likelihood of LULC changes). The approach employed in this study is particularly valuable in a situation like Ethiopia and other countries with similar conditions where there are complex processes behind the dynamics of LULC types. The identified drivers can be used as inputs for modelling of future changes. Furthermore, evidence from this study indicated that in response to the drivers, local people amplified the LULC dynamics while using the resources as a means of survival or source of income for their livelihoods. The same trends are expected to happen in the Ethiopian highlands of similar settings where subsistence farming is still present. Thus, appropriate strategy centered on those major drivers is urgently required in order to avert the ongoing undesirable LULC changes of resources.

The LULC changes over time have not been beneficial to the ecosystem services. This was revealed by the analysis of changes of ESVs in response to the four decades of changes in LULC types in the studied landscape. The estimate showed that LULC dynamics between 1973 and 2012 in the studied landscape, which covers about 1091 km^2, have resulted in a loss of ecosystem services ranging from US$ 19.3 million per year, when using own modified more conservative value coefficients, to US$ 45.9 million per year when using global value coefficients. The identified loss of ecosystem service values were, mainly, linked to the huge conversion of forests, which were the main providers of ecosystem services. The present study provided a fast and effective way to assess the changes of ESVs. The use of GIS and datasets derived from remotely sensed imagery, with their corresponding value coefficients, as a proxy of measurement facilitated the estimation process. Although the considered LULC datasets have the highest quality in terms of accuracy, the corresponding value coefficients were rough and the results of this study must be considered as minimum estimates. Apart from this, evidence from the landscape level estimate indicated that the decline of ESVs reflected the effects of ecological degradation to some extent. Thus, the outputs can be used as references and the bases for stimulating discussions during policy formulations that can improve decisions involving various aspects of intervention strategies to maintain a balance between economic development and ecosystem health in the future. The economic gains that are aimed to be achieved under the current patterns of agricultural development in the studied landscape as well as from other area of similar setting should consider the non-market economic losses that occur as natural ecosystems (e.g. natural forests and woodlands) are lost or transformed. To enhance the credibility of valuation for practically applicable local level policy formulation, regarding alternative land resource options, it is important to obtain value coefficients for ecosystem service values that reflect local conditions and recalculating them using the approach presented in this study.

This study also revealed different possible future LULC patterns and changes depending on the considered three scenarios across a range of potential social, environmental and economic policy contexts. In this regard, the study disclosed the ability of GIS to successfully simulate LULC patterns and changes for the first time in the studied landscape of the Ethiopian highlands. However, a further study is suggested to compare the considered scenarios with mechanistic LULC modelling approach that have high potential to investigate the problem from a financial policies point of view.

The simulation results indicated that croplands will expand widely in the study landscape under the BAU scenario and would expand to the remaining woodlands, natural forests and grasslands, reflecting vulnerability of these LULC types and potential loss of associated ESVs in the future. Therefore, this scenario is not suitable for sustainable resource use and conservation, highlighting the need to seriously consider such undesirable LULC developments, particularly expansion of croplands in future land use planning processes. Under the FCWP scenario, attempting only strict implementation of current protection and conservation policies will bring competition among other LULC types, particularly more pressure will be extended on the grassland ecosystem. According to the simulation outputs of this scenario, the grassland ecosystem that, even, occupy steep and very steep slopes will also be converted to croplands. As a result, this scenario will lead to serious environmental crises as well. The SI scenario, with holistic approach, demonstrated that expansion of croplands could vigorously be reduced, remaining forests would be better conserved and degraded land would be recovered, resulting in gains of the associated total ESVs. Consequently, the results of this scenario have implications for better landscape management by ensuring expected agricultural production while safeguarding the environment. Thus, the SI scenario confirmed the need for integrated landscape management approach, i.e. simultaneous implementation of better family planning and improved farming system to a target landscape that are critical for a sustainable future in the Ethiopian highlands beyond just conservation and protection policies.

The modelling approach framed in this study is a useful tool and, particularly, valuable in a situation like Ethiopia, and other countries with similar conditions where there are complex processes that need to be considered for sustainable management of resources. It is flexible, spatially explicit and scenario-based, and has tremendous values for education and research applications related to future LULC change. The approach can also be used to identify LULC types that may deserve priority attention and enhance understanding of the potential effect of attempting only strict implementation of conservation and protection policies without considering other important issues that have consequences. Local authorities can understand the complex LULC system better by using the results of this study. The results are also expected to provoke public awareness on the state of natural resources and benefit from the developed framework for appropriate management options. A further study is suggested to practically test the presented modelling framework through research-for-development approach in a test site, preferably, in the studied landscape or other areas with similar settings so that it can be used as a model area for effective use and conservation of the natural resources.

References

Abate, A. 2004. Biomass and Nutrient Studies of Selected Tree Species of Natural and Plantation Forests. Implications for a Sustainable Management of the Munessa-Shashemene Forest, Ethiopia. Universität Bayreuth, Bayreuth, 149 pp.

Abate, S. 1994. Land use dynamics, soil degradation, and potential for sustainable use in Metu area, Illubabor Region, Ethiopia. University of Berne Switzerland, Institute of Geography, Berne, Switzerland, 135 pp.

Abebaw, D., Fentie, Y., Kassa, B. 2010. The impact of a food security program on household food consumption in Northwestern Ethiopia: A matching estimator approach. Food Policy 35, 286–293.

Adimassu, Z., Gorfu, B., Nigussie, D., Mowo, J., Hilemichael, K. 2013. Farmers' Preference for Soil and Water Conservation Practices in Central Highlands of Ethiopia. Afr. Crop Sci. J. 21, 781–790.

Alemayehu, T., Furi, W., Legesse, D. 2007. Impact of water overexploitation on highland lakes of eastern Ethiopia. Environ Geol. 52, 147–154.

Allan, J.D. 2004. Landscapes and Riverscapes. The Influence of Land Use on Stream Ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257–284.

Amessie, G. 2014. Opportunities and Challenges of Community Based Natural Resource Management in Menz-Guassa Community Conservation Area, Ethiopia, MA. Thesis, Addis Ababa, Ethiopia, 83 pp.

Anderson, J.R., Hardy, E.E., Roach, J.T., Witmer, R.E. 1976. A land use and land cover classification system for use with remote sensor data. US Geological Survey Professional Paper, No. 964, Reston, Virginia, USA. 28 pp.

Aran, D., Gury, M., Jeanroy, E. 2001. Organo-metallic complexes in an Andosol: a comparative study with a Cambisol and Podzol. Geoderma 99, 65–79.

Arora, V. 2002. Modeling vegetation as a dynamic component in soil-vegetation-atmosphere transfer schemes and hydrological models. Rev. Geophys. 40, 1–25.

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.

Asfaw, A., Demissie, Y. 2012. Sustainable Household Energy for Addis Ababa, Ethiopia. Consilience: The J. Sustain. Develop. 8, 1–11.

Assen, M. 2011. Land use/cover dynamics and its implications in the dried Lake Alemaya watershed, eastern Ethiopia. J. Sustain. Dev. Africa 13, 267–284.

Avelar, S., Zah, R., Tavares-Corrêa, C. 2009. Linking socioeconomic classes and land cover data in Lima, Peru. Assessment through the application of remote sensing and GIS. Int. J. Appl. Earth Obs. Geoinformation 11, 27–37.

Baatz, M., Schäpe, A. 2000. Multiresolution Segmentation: an optimization approach for high quality multi-scale image. In: Blaschke, T., Griesebner, G., Strobl, J. (eds.), Angewandte geographische Informationsverarbeitung XII. Beiträge zum AGIT-Symposium Salzburg 2000. Herbert Wichmann Verlag, Heidelberg, pp. 12–23.

Bachelet, D., Neilson, R.P., Hickler, T., Drapek, R.J., Lenihan, J.M., Sykes, M.T., Smith, B., Sitch, S., Thonicke, K. 2003. Simulating past and future dynamics of natural ecosystems in the United States. Glob. Biogeochem. Cycles 17, 1045.

Baker, W.L. 1989. A review of models of landscape change. Landscape Ecol. 2, 111–133.

Baron, J.S., Hartman, M.D., Kittel, T.G.F., Band, L.E., Ojima, D.S., Lammers, R.B. 1998. Effects of Land Cover, Water Redistribution, and Temperature on Ecosystem Processes in the South Platte Basin. Econ. Appl. 8, 1037–1051.

Barral, M.P., Oscar, M.N. 2012. Land-use planning based on ecosystem service assessment: A case study in the Southeast Pampas of Argentina. Agric. Ecosyst. Environ. 154, 34–43.

Bateman, I.J., Harwood, A.R., Mace, G.M., Watson, R.T., Abson, D.J., Andrews, B., Binner, A., Crowe, A., Day, B.H., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Lovett, A.A., Munday, P., Pascual, U., Paterson, J., Perino, G., Sen, A., Siriwardena, G., van Soest, D., Termansen, M. 2013. Bringing Ecosystem Services into Economic Decision-Making: Land Use in the United Kingdom. Science 341 (6141), 45–50.

Baudron, F., Corbeels, M., Andersson, J.A., Sibanda, M., Giller, K.E. 2011. Delineating the drivers of waning wildlife habitat: The predominance of cotton farming on the fringe of protected areas in the Mid-Zambezi Valley, Zimbabwe. Biol. Conserv. 144, 1481–1493.

Bax, V., Francesconi, W., Quintero, M. 2016. Spatial modeling of deforestation processes in the Central Peruvian Amazon. J. Nat. Conserv. 29, 79–88.

Beilin, R., Lindborg, R., Stenseke, M., Pereira, H.M., Llausàs, A., Slätmo, E., Cerqueira, Y., Navarro, L., Rodrigues, P., Reichelt, N., Munro, N., Queiroz, C. 2014. Analysing how drivers of agricultural land abandonment affect biodiversity and cultural landscapes using case studies from Scandinavia, Iberia and Oceania. Land Use Policy 36, 60–72.

Belal, A., Moghanm, F. 2011. Detecting urban growth using remote sensing and GIS techniques in Al Gharbiya governorate, Egypt. Egypt. J. Remote Sens. Space Sci. 14, 73–79.

Belward, A.S., Skøien, J.O. 2015. Who launched what, when and why; trends in global land-cover observation capacity from civilian earth observation satellites. ISPRS J. Photogramm. Remote Sens. 103, 115–128.

Beniston, M. 2003. Climate Change in Mountain Regions: A review of possible impacts. Clim. Chang. 59, 5–31.

Bewket, W. 2002. Land Cover Dynamics Since the 1950s in Chemoga Watershed, Blue Nile Basin, Ethiopia. Mt. Res. Dev. 263–269.

Bewket, W., Conway, D. 2007. A note on the temporal and spatial variability of rainfall in the drought-prone Amhara region of Ethiopia. Int. J. Climatol.,vol. 27, 1467–1477.

Bhattacharjee, S., Ghosh, S.K. 2015. Spatio-Temporal Change Modeling of LULC: A Semantic Kriging Approach. ISPRS Ann. Photogramm. Remote Sens. Spatial Inf. Sci. 2, 177–184.

Biazin, B., Sterk, G. 2013. Drought vulnerability drives land-use and land cover changes in the Rift Valley dry lands of Ethiopia. Agric. Ecosyst. Environ. 164, 100–113.

Bishaw, B. 2009. Deforestation and Land Degradation in the Ethiopian Highlands: A Strategy for Physical Recovery. E. e-J. Res. Innov. 1, 5–18.

Blaschke, T. 2010. Object based image analysis for remote sensing. ISPRS J. Photogramm. Remote Sens. 65, 2–16.

Braimoh, A.K. 2004. Seasonal migration and land-use change in Ghana. Land Degrad. Dev. 15, 37–47.

Brink, A.B., Bodart, C., Brodsky, L., Defourney, P., Ernst, C., Donney, F., Lupi, A., Tuckova, K. 2014. Anthropogenic pressure in East Africa—Monitoring 20 years of land cover changes by means of medium resolution satellite data. Int. J. Appl. Earth Obs. Geoinf. 28, 60–69.

Brockerhoff, E.G., Jactel, H., Parrotta, J.A., Quine, C.P., Sayer, J. 2008. Plantation forests and biodiversity: oxymoron or opportunity? Biodivers. Conserv. 17, 925–951.

Brown, D.G., Walker, R., Manson, S., Seto, K. 2004. Modeling Land Use and Land Cover Change. In: Gutman, G, Janetos, A, Justice, C, Moran, E, Mustard, J, Rindfuss, R, Skole, D, Turner II, B. L (eds.), Land Change Science: Observing, Monitoring, and Understanding Trajectories of Change on the Earth's Surface. Kluwer Academic, Dordrecht, The Netherlands, pp. 395–409

Bürgi, M., Hersperger, A.M., Schneeberger, N. 2005. Driving forces of landscape change - current and new directions. Landscape Ecol. 19, 857–868.

Burrough, P. 1986. Principles of geographical information systems for land resources assessment. Geocarto International 1, 54–54.

Butler, J.R., Wong, G.Y., Metcalfe, D.J., Honzák, M., Pert, P.L., Rao, N., van Grieken, M.E., Lawson, T., Bruce, C., Kroon, F.J., Brodie, J.E. 2013. An analysis of trade-offs between multiple ecosystem services and stakeholders linked to land use and water quality management in the Great Barrier Reef, Australia. Ecosyst. Environ. 180, 176–191.

Campbell, D.J., Lusch, D.P., Smucker, T.A., Wangui, E.E. 2005. Multiple Methods in the Study of Driving Forces of Land Use and Land Cover Change: A Case Study of SE Kajiado District, Kenya. Hum Ecol. 33, 763–794.

Carver, S., Comber, A., McMorran, R., Nutter, S., 2012. A GIS model for mapping spatial patterns and distribution of wild land in Scotland. Landsc. Urban Plan. 104, 395–409.

Cerbu, G.A., Sonwa, D.J., Pokorny, B. 2013. Opportunities for and capacity barriers to the implementation of REDD+ projects with smallholder farmers: Case study of Awae and Akok, Centre and South Regions, Cameroon. For. Policy Econ. 36, 60–70.

Chaffey, D. 1980. South-west Ethiopia forest inventory project: An inventory of forest at Munessa and Shashamene, London.

Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C., Daily, G.C. 2006. Conservation Planning for Ecosystem Services. Plos Biol. 4, 2138–2152

Chen, H., Pontius, R.G. 2010. Diagnostic tools to evaluate a spatial land change projection along a gradient of an explanatory variable. Landsc. Ecol. 25, 1319–1331.

Chopra, K. 1993. The Value of Non-Timber Forest Products: An Estimation for Tropical Deciduous Forests in India. Econ. Bot. 47, 251–257.

Congalton, R.G., Green, K. 2009. Assessing the accuracy of remotely sensed data. Principles and practices. CRC Press/Taylor & Francis, Boca Raton, 183 pp.

Coria, J., Sterner, T. 2011. Natural Resource Management: Challenges and Policy Options. Annu. Rev. Resour. Econ. 3, 203–230.

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

Costanza, R., Groot, R. de, Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S., Turner, R.K. 2014. Changes in the global value of ecosystem services. Glob. Environ. Chang. 26, 152–158.

Creswell, J.W., 2014. Research design. Qualitative, quantitative, and mixed methods approaches. SAGE Publications, Thousand Oaks, California, 342pp.

Central Statistical Authority (CSA). 2007. Summary and Statistical Report of the 2007 Population and Housing Census, Addis Ababa.

Central Statistical Authority (CSA). 2013. Population Projection of Ethiopia for All Regions at Wereda Level from 2014 – 2017, Addis Ababa.

Cyranoski, D. 2007. Biodiversity: Logging: the new conservation. Nature 446, 608–610.

Czajkowski, M., Ahtiainen, H., Artell, J., Meyerhoff, J. 2017. Choosing a functional form for an international benefit transfer: Evidence from a nine-country valuation experiment. Ecol. Econ. 134, 104-113

Daba, S. 2003. An investigation of the physical and socioeconomic determinants of soil erosion in the Hararghe Highlands, eastern Ethiopia. Land Degrad. Dev. 14, 69–81.

Daily, G.C., Söderqvist, T., Aniyar, S., Arrow, K., Dasgupta, P., Ehrlich, P.R., Folke, C., Jansson, A., Jansson, B.-O., Kautsky, N., Levin, S., Lubchenco, J., Mäler, K.-G., Simpson, D., Starrett, D., Tilman, D., Walker, B., 2000. The Value of Nature and the Nature of Value. Science 289, 395–396.

Dallimer, M., Davies, Z.G., Diaz-Porras, D.F., Irvine, K.N., Maltby, L., Warren, P.H., Armsworth, P.R., Gaston, K.J. 2015. Historical influences on the current provision of multiple ecosystem services. Global Environ. 31, 307–317.

Davis, J., O'Grady, A.P., Dale, A., Arthington, A.H., Gell, P.A., Driver, P.D., Bond, N., Casanova, M., Finlayson, M., Watts, R.J., Capon, S.J., Nagelkerken, I., Tingley, R., Fry, B., Page, T.J., Specht, A. 2015. When trends intersect: The challenge of protecting freshwater ecosystems under multiple land use and hydrological intensification scenarios. Sci. Total Environ. 534, 65–78.

Dayamba, S.D., Djoudi, H., Zida, M., Sawadogo, L., Verchot, L. 2016. Biodiversity and carbon stocks in different land use types in the Sudanian Zone of Burkina Faso, West Africa. Agri. Ecosy. Environ. 216, 61–72.

de Groot, R.S., Wilson, M.A., Boumans, R.M. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol. Econ. 4, 393–408.

de Souza, A.L., Fonseca, D.G., Libório, R.A., Tanaka, M.O. 2013. Influence of riparian vegetation and forest structure on the water quality of rural low-order streams in SE Brazil. For. Ecol. Manage. 298, 12–18.

Definiens. 2009. eCognition Developer 8 Reference Book User Guide, München, Germany.

Dercon, S. 1999. Who benefits from good weather and reforms? A Study of Ethiopian Villages. In: CSAE (ed.), Paper presented at the conference on Poverty in Africa: a Dialogue on Causes and Solutions.

Dessie, G., Christiansson, C. 2008. Forest Decline and Its Causes in the South-Central Rift Valley of Ethiopia: Human Impact over a One Hundred Year Perspective. Ambio 37, 263–271.

Dessie, G., Kleman, J. 2007. Pattern and Magnitude of Deforestation in the South Central Rift Valley Region of Ethiopia. Mt. Res. Dev. 27, 162–168.

Dietrich, J.P., Schmitz, C., Lotze-Campen, H., Popp, A., Müller, C. 2014. Forecasting technological change in agriculture—An endogenous implementation in a global land use model. Technol. Forecast. Soc. l. 81, 236–249.

Dronova, I., Gong, P., Wang, L. 2011. Object-based analysis and change detection of major wetland cover types and their classification uncertainty during the low water period at Poyang Lake, China. Remote Sens. Environ. 115, 3220–3236.

Dwivedi, R.S., Sreenivas, K., Ramana, K.V. 2005. Cover: Land-use/land-cover change analysis in part of Ethiopia using Landsat Thematic Mapper data. Int. J. Remote Sens. 26, 1285–1287.

Echeverria, C., Coomes, D.A., Hall, M., Newton, A.C. 2008. Spatially explicit models to analyze forest loss and fragmentation between 1976 and 2020 in southern Chile. Ecol. Model. 212, 439–449.

Ethiopian Mapping Authority (EMA). 1988. National Atlas of Ethiopia, Addis Ababa.

Environmental Protection Authority (EPA). 1997. Conservation Strategy of Ethiopia, Addis Ababa.

Eshetu, Z., Högberg, P. 2000. Reconstruction of Forest Site History in Ethiopian Highlands Based on 13 C Natural Abundance of Soils. Ambio 29, 83–89.

Environmental System Research Institute (Esri). 2011. GIS Best Practices. GIS for Urban and Regional Planning, Redlands, CA, 68 pp.

Estes, A.B., Kuemmerle, T., Kushnir, H., Radeloff, V.C., Shugart, H.H. 2012. Land-cover change and human population trends in the greater Serengeti ecosystem from 1984–2003. Biol. Conserv. 147, 255–263.

Estifanos, T.K. 2008. Integrated Assessment of ecosystem services and stakeholder analysis of Abijata-Shalla Lakes National Park, Ethiopia, MSc Thesis, Wageningen, The Netherlands, 150 pp.

Estrada-Carmona, N., Hart, A.K., DeClerck, F.A., Harvey, C.A., Milder, J.C. 2014. Integrated landscape management for agriculture, rural livelihoods, and ecosystem conservation: An assessment of experience from Latin America and the Caribbean. Landsc. Urban Plan.129, 1–11.

Ethiopian Ministry of Finance and Economic Development (MoFED). 2004. Sustainable development and poverty reduction program (SDPRP). Annual Progress Report 2003/04.

Ethiopian Ministry of Finance and Economic Development (MoFED). 2006. Building on progress: A plan for accelerated and sustained development to end poverty (PASDEP). Annual Progress Report 2005/06.

Ethiopian Ministry of Finance and Economic Development (MoFED). 2010. Growth and transformation plan of Ethiopia (GTP).

Fairman, J.G., Nair, U.S., Christopher, S.A., Mölg, T. 2011. Land use change impacts on regional climate over Kilimanjaro. J. Geophys. Res.116.

Fan, F., Weng, Q., Wang, Y., 2007. Land Use and Land Cover Change in Guangzhou, China, from 1998 to 2003, Based on Landsat TM /ETM+ Imagery. Sensors 7, 1323–1342.

FAO, 1990. Guidelines for Soil description, Rome.

Federal Democratic Republic of Ethiopia (FDRE). 2000. Ethiopian Water Resource Management Proclamation No. 197/2000, Berhan ena Selam Printing Press, Addis Ababa, Ethiopia.

Federal Democratic Republic of Ethiopia (FDRE). 2007. Forest Development, Conservation and Utilization Proclamation No. 542/2007, Berhan ena Selam Printing Press, Addis Ababa, Ethiopia.

Fiquepron, J., Garcia, S., Stenger, A. 2013. Land use impact on water quality: Valuing forest services in terms of the water supply sector. J. Environ. Manage.126, 113–121.

Flanders, D., Hall-Beyer, M., Pereverzoff, J. 2003. Preliminary evaluation of eCognition object-based software for cut block delineation and feature extraction. Can. J. Remote Sens. 29, 441–452.

Foody, G.M. 2001. Monitoring the magnitude of land-cover change around the southern limits of the Sahara. Photogramm. Eng. Remote Sens. 67, 841–847.

Foody, G.M. 2002. Status of land cover classification accuracy assessment. Remote Sens. Environ. 80, 185–201.

Frélichová, J., Vačkář, D., Pártl, A., Loučková, B., Harmáčková, Z.V., Lorencová, E. 2014. Integrated assessment of ecosystem services in the Czech Republic. Ecosyst. Serv. 8, 110–117.

Fritzsche, F., Zech, W., Guggenberger, G. 2007. Soils of the Main Ethiopian Rift Valley escarpment: A transect study. Catena 70, 209–219.

Garedew, E., Sandewall, M., Soderberg, U. 2012. A Dynamic Simulation Model of Land-Use, Population, and Rural Livelihoods in the Central Rift Valley of Ethiopia. Environ. Manage. 49, 151–162.

Gautam, N., Chennaiah, G. 1985. Land-use and land-cover mapping and change detection in Tripura using satellite LANDSAT data. Int. J. Remote Sens. 6, 517–528.

Gillanders, S.N., Coops, N.C., Wulder, M.A., Gergel, S.E., Nelson, T. 2008. Multi-temporal remote sensing of landscape dynamics and pattern change: describing natural and anthropogenic trends. Progr. Phys. Geogr. 32, 503–528.

Gilligan, D.O., Hoddinott, J., Taffesse, A.S. 2009. The Impact of Ethiopia's Productive Safety Net Programme and its Linkages. J. Dev. Stud. 45, 1684–1706.

Giri, C., Zhu, Z., Reed, B. 2005. A comparative analysis of the Global Land Cover 2000 and MODIS land cover data sets. Remote Sens. Environ. 94, 123–132.

Grinblat, Y., Kidron, G.J., Karnieli, A., Benenson, I. 2015. Simulating land-use degradation in West Africa with the ALADYN model. J. Arid Environ.112, 52–63.

Guo, Z., Xiao, X., Gan, Y., Zheng, Y. 2001. Ecosystem functions, services and their values – a case study in Xingshan County of China. Ecol. Econ. 38, 141–154.

Guo, Z., Zhang, L., Li, Y., Romanuk, T.N. 2010. Increased Dependence of Humans on Ecosystem Services and Biodiversity. PLoS ONE 5, e13113.

Hamandawana, H., Nkambwe, M., Chanda, R., Eckardt, F. 2005. Population driven changes in land use in Zimbabwe's Gutu district of Masvingo province: Some lessons from recent history. Applied Geogra. 25, 248–270.

Hayase, Y., Liaw, K.-L. 1997. Factors on Polygamy in Sub-Saharan Africa: Findings Based on The Demographic and Health Surveys. The Developing Econ. 35, 293–327.

Hein, L., van Koppen, K., Groot, R.S. de, van Ierland, E.C. 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecol. Econ. 57, 209–228.

Hietel, E., Waldhardt, R., Otte, A. 2004. Analysing land-cover changes in relation to environmental variables in Hesse, Germany. Landscape Ecol. 19, 473–489.

Higgins, S.I., Turpie, J.K., Costanza, R., Cowling, R.M., Le Maitre, D.C., Marais, C., Midgley, G.F. 1997. An ecological economic simulation model of mountain fynbos ecosystems. Ecol. Econ. 22, 155–169.

Hijmans, R.J., Cameron, S.E., Parra, J.L., Jones, P.G., Jarvis, A. 2005. Very high resolution interpolated climate surfaces for global land areas. Int. J. Climatol. 25, 1965–1978.

Holden, S., Shiferaw, B. 2004. Land degradation, drought and food security in a less-favoured area in the Ethiopian highlands: a bio-economic model with market imperfections. Agric. Econ. 30, 31–49.

Hope, C., Maul, P. 1996. Valuing the impact of CO2 emissions. Energy Policy 24, 211–219.

Hu, H., Liu, W., Cao, M. 2008. Impact of land use and land cover changes on ecosystem services in Menglun, Xishuangbanna, Southwest China. Monit. Assess. 146, 147–156.

Hurni, H. 1998. Agroecological Belts of Ethiopia: Explanatory notes on three maps at a scale of 1:1,000,000, Addis Ababa.

Hurni, H., Tato, K., Zeleke, G. 2005. The Implications of Changes in Population, Land Use, and Land Management for Surface Runoff in the Upper Nile Basin Area of Ethiopia. Mt. Res. Dev. 25, 147–154.

Jackson, B., Pagella, T., Sinclair, F., Orellana, B., Henshaw, A., Reynolds, B., Mcintyre, N., Wheater, H., Eycott, A. 2013. Polyscape: A GIS mapping framework providing efficient and spatially explicit landscape-scale valuation of multiple ecosystem services. Landsc. Urban Plann. 112, 74–88.

Jadin, I., Vanacker, V., Hoang, H.T.T. 2013. Drivers of Forest Cover Dynamics in Smallholder Farming Systems: The Case of Northwestern Vietnam. Ambio 42, 344–356.

Juutinen, A., Kosenius, A.-K., Ovaskainen, V. 2014. Estimating the benefits of recreation-oriented management in state-owned commercial forests in Finland: A choice experiment. J. For. Econ. 20, 396–412.

Kamusoko, C., Oono, K., Nakazawa, A., Wada, Y., Nakada, R., Hosokawa, T., Tomimura, S., Furuya, T., Iwata, A., Moriike, H., Someya, T., Yamase, T., Nasu, M., Gomi, Y., Sano, T., Isobe, T., Homsysavath, K. 2011. Spatial Simulation Modelling of Future Forest Cover Change Scenarios in Luangprabang Province, Lao PDR. Forests 2, 707–729.

Kassahun, F. 2011. Modelling Informal Settlement Growth in Dar es Selaam, Tanzania. University of Twenete, Enschede, The Netherlands, 110 pp.

Kassie, M., Zikhali, P., Pender, J., Köhlin, G. 2010. The Economics of Sustainable Land Management Practices in the Ethiopian Highlands. J. Agr. Econ. 61, 605–627.

Kicklighter, D.W., Cai, Y., Zhuang, Q., Parfenova, E.I., Paltsev, S., Sokolov, A.P., Melillo, J.M., Reilly, J.M., Tchebakova, N.M., Lu, X. 2014. Potential influence of climate-induced vegetation shifts on future land use and associated land carbon fluxes in Northern Eurasia. Environ. Res. Lett. 9, 35004.

Kidane-Mariam, T. 2015. Ethiopia: Opportunities and Challenges of Tourism Development in the Addis Ababa-upper Rift Valley Corridor. J. Tour. Hospi. 167.

Kindu, M., Schneider, T., Teketay, D., Knoke, T. 2013. Land Use/Land Cover Change Analysis Using Object-Based Classification Approach in Munessa-Shashemene Landscape of the Ethiopian Highlands. Remote Sens. 5, 2411–2435.

Kindu, M., Schneider, T., Teketay, D., Knoke, T. 2015. Drivers of land use/land cover changes in Munessa-Shashemene landscape of the south-central highlands of Ethiopia. Environ Monit Assess. 7, 452.

Kindu, M., Schneider, T., Teketay, D., Knoke, T. 2016. Changes of ecosystem service values in response to land use/land cover dynamics in Munessa–Shashemene landscape of the Ethiopian highlands. Sci. Total Environ. 547, 137–147.

Knoke, T., Bendix, J., Pohle, P., Hamer, U., Hildebrandt, P., Roos, K., Gerique, A., Sandoval, M.L., Breuer, L., Tischer, A., Silva, B., Calvas, B., Aguirre, N., Castro, L.M., Windhorst, D., Weber, M., Stimm, B., Günter, S., Palomeque, X., Mora, J., Mosandl, R., Beck, E. 2014. Afforestation or intense pasturing improve the ecological and economic value of abandoned tropical farmlands. Nat Comms. 5, 5612.

Knoke, T., Calvas, B., Moreno, S.O., Onyekwelu, J.C., Griess, V.C. 2013. Food production and climate protection—What abandoned lands can do to preserve natural forests. Glob. Environ. Chang. 23, 1064–1072.

Knoke, T., Hahn, A. 2013. Global Change and the Role of Forests in Future Land-Use Systems. In: Matyssek, R., Clarke, N., Cudlin, P., Mikkelsen, T., Tuovinen, J.-P., Wieser, G., Paoletti, E. (eds.), Climate Change, Air Pollution and Global Challenges. Elsevier, pp. 569–588.

Knoke, T., Paul, C., Härtl, F., Castro, L.M., Calvas, B., Hildebrandt, P. 2015. Optimizing agricultural land-use portfolios with scarce data—A non-stochastic model. Ecol. Econ. 120, 250–259.

Knoke, T., Paul, C., Hildebrandt, P., Calvas, B., Castro, L.M., Härtl, F., Döllerer, M., Hamer, U., Windhorst, D., Wiersma, Y.F., Curatola Fernández, G.F., Obermeier, W.A., Adams, J., Breuer, L., Mosandl, R., Beck, E., Weber, M., Stimm, B., Haber, W., Fürst, C., Bendix, J. 2016. Compositional diversity of rehabilitated tropical lands supports multiple ecosystem services and buffers uncertainties. Nat Comms. 7, 11877.

Knoke, T., Román-Cuesta, R.M., Weber, M., Haber, W. 2012. How can climate policy benefit from comprehensive land-use approaches? Front. Ecol. Environ. 10, 438–445.

Knoke, T., Steinbeis, O.-E., Bösch, M., Román-Cuesta, R.M., 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.

Konarska, K.M., Sutton, P.C., Castellon, M. 2002. Evaluating scale dependence of ecosystem service valuation: a comparison of NOAA-AVHRR and Landsat TM datasets. Ecol. Econ. 41, 491–507.

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

Kubiszewski, I., Costanza, R., Dorji, L., Thoennes, P., Tshering, K. 2013. An initial estimate of the value of ecosystem services in Bhutan. Ecosyst. Serv. 3, e11–e21.

La Rosa, D. 2014. Accessibility to greenspaces: GIS based indicators for sustainable planning in a dense urban context. Ecol. Indic. 42, 122–134.

Lambin, E.F., Geist, H. 2006. Land-use and land-cover change. Local processes and global impacts. Springer, Berlin, New York, 222pp.

Lambin, E.F., Geist, H.J., Lepers, E. 2003. Dynamics of Land Use and Land Cover Change in Tropical Regions. Annu. Rev. Environ. Resourc. 28, 205–241.

Latinopoulos, D., Kechagia, K. 2015. A GIS-based multi-criteria evaluation for wind farm site selection. A regional scale application in Greece. Renew. Energy 78, 550–560.

Leh, M.D., Matlock, M.D., Cummings, E.C., Nalley, L.L. 2013. Quantifying and mapping multiple ecosystem services change in West Africa. Ecosyst. Environ. 165, 6–18.

Lemenih, M. 2004. Effects of land use changes on soil quality and native flora degradation and restoration in the highlands of Ethiopia. Implication for Sustainable Land Management. Ph.D. Thesis, Swedish University of Agricultural Sciences, Umeå.

Lemenih, M., Karltun, E., Olsson, M. 2005. Assessing soil chemical and physical property responses to deforestation and subsequent cultivation in smallholders farming system in Ethiopia. Agric. Ecosyst. Environ. 105, 373–386.

Lemenih, M., Kassa, H. 2011. Opportunities and challenges for sustainable production and marketing of gums and resins in Ethiopia. Center for International Forestry Research, Bogor, Indonesia, 93 pp.

Lemenih, M., Teketay D. 2004a. Potentials of plantation forests in fostering the restoration of native flora and fauna at degraded sites in Ethiopia: A Review. Ethiop. J. Biol. Soc. 3, 81–111.

Lemenih, M., Teketay, D. 2004b. Restoration of Native Forest Flora in the Degraded Highlands of Ethiopia: Constraints and Opportunities. SINET: Ethiop. J. Sci. 27, 75–90.

Lemenih, M., Teketay, D. 2005. Effect of prior land use on the recolonization of native woody species under plantation forests in the highlands of Ethiopia. For. Ecol. Manage. 218, 60–73.

Lesschen, J.P., Verburg, P.H., Staal, S.J. 2005. Statistical methods for analysing the spatial dimension of changes in land use and farming systems. International Livestock Research Institute; Wageningen University, LUCC Focus 3 Office, Nairobi, Wageningen, 80 pp.

Li, J., Wang, W., Hu, G., Wei, Z. 2010. Changes in ecosystem service values in Zoige Plateau, China. Agric. Ecosyst. Environ. 139, 766–770.

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.

Lillesand, T.M., Kiefer, R.W. 2000. Remote sensing and image interpretation. John Wiley & Sons, New York, USA, 724pp.

Limburg, K.E., O'Neill, R.V., Costanza, R., Farber, S. 2002. Complex systems and valuation. Ecol. Econ. 41, 409–420.

Liu, S., Costanza, R., Troy, A., D'Aagostino, J., Mates, W. 2010. Valuing New Jersey's Ecosystem Services and Natural Capital: A Spatially Explicit Benefit Transfer Approach. Environ. Manage. 45, 1271–1285.

Liu, Y., Li, J., Zhang, H. 2012. An ecosystem service valuation of land use change in Taiyuan City, China. Ecol. Mod. 225, 127–132.

Long, H., Wu, X., Wang, W., Dong, G. 2008. Analysis of Urban-Rural Land-Use Change during 1995-2006 and Its Policy Dimensional Driving Forces in Chongqing, China. Sensors 8, 681–699.

Lu, D., Mausel, P., Brondízio, E., Moran, E. 2004. Change detection techniques. Int. J. Remote Sens. 25, 2365–2401.

Luisetti, T., Turner, R.K., Jickells, T., Andrews, J., Elliott, M., Schaafsma, M., Beaumont, N., Malcolm, S., Burdon, D., Adams, C., Watts, W., 2014. Coastal Zone Ecosystem Services: from science to values and decision making; a case study. Sci. Total Environ. 493, 682–693.

Lundgren, B. 1971. Soil studies in a montane forest in Ethiopia. Research Notes Nr 11., Stockholm.

Lunetta, R.S., Knight, J.F., Ediriwickrema, J., Lyon, J.G., Worthy, L.D. 2006. Land-cover change detection using multi-temporal MODIS NDVI data. Remote Sens. Environ. 105, 142–154.

Luo, L., Mountrakis, G. 2011. Converting local spectral and spatial information from a priori classifiers into contextual knowledge for impervious surface classification. ISPRS J. Photogramm. Remote Sens. 66, 579–587.

Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J.P., Grizzetti, B., Drakou, E.G., La Notte, A., Zulian, G., Bouraoui, F., Luisa Paracchini, M., Braat, L., Bidoglio, G. 2012. Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst. Serv. 1, 31–39.

Manakos, I., Schneider, T., Ammer, U. 2000. A comparison between the ISODATA and the eCognition classification methods on basis of field data; In Proceedings of the XIXth ISPRS Congress, Amsterdam, The Netherlands.

Manrique, S., Franco, J., Núñez, V., Seghezzo, L. 2011. Potential of native forests for the mitigation of greenhouse gases in Salta, Argentina. Biomass Bioenergy 35, 2184–2193.

Martínez, J.-M.Á., Suárez-Seoane, S., Luis Calabuig, E. de. 2011. Modelling the risk of land cover change from environmental and socio-economic drivers in heterogeneous and changing landscapes: The role of uncertainty. Landscape Urban Plann. 101, 108–119.

Martínez, M.L., Pérez-Maqueo, O., Vázquez, G., Castillo-Campos, G., García-Franco, J., Mehltreter, K., Equihua, M., Landgrave, R. 2009. Effects of land use change on biodiversity and ecosystem services in tropical montane cloud forests of Mexico. For. Ecol. Manage. 258, 1856–1863.

Martinuzzi, S., Radeloff, V.C., Joppa, L.N., Hamilton, C.M., Helmers, D.P., Plantinga, A.J., Lewis, D.J. 2015. Scenarios of future land use change around United States' protected areas. Biol. Conserv. 184, 446–455.

Mas, J.-F., Velázquez, A., Díaz-Gallegos, J.R., Mayorga-Saucedo, R., Alcántara, C., Bocco, G., Castro, R., Fernández, T., Pérez-Vega, A. 2004. Assessing land use/cover changes: a nationwide multidate spatial database for Mexico. Int. J. Appl. Earth Obs. Geoinf. 5, 249–261.

McCann, J. 1995. People of the plow. An agricultural history of Ethiopia, 1800-1990. University of Wisconsin Press, Madison, 298 pp.

Millennium Ecosystem Assessment (MEA). 2005. Ecosystems and human well-being. Synthesis. Island Press, Washington, DC, 137pp.

Messina, J.P., Walsh, S.J. 2001. 2.5DMorphogenesis: modeling landuse and landcover dynamics in the Ecuadorian Amazon. Plant Ecol. 156, 75–88.

Meyer, W.B., Turner, B.L. 1992. Human population growth and global land-use/cover change. Ann. Rev. Ecol. Syst. 23, 39–61.

Ministry of Agriculture (MOA). 2000. Agro-Ecological Zones of Ethiopia, Addis Ababa.

Mottet, A., Ladet, S., Coqué, N., Gibon, A. 2006. Agricultural land-use change and its drivers in mountain landscapes: A case study in the Pyrenees. Agric. Ecosyst. Environ. 114, 296–310.

Nagendra, H., Southworth, J., Tucker, C. 2003. Accessibility as a determinant of landscape transformation in western Honduras: linking pattern and process. Landsc. Ecol. 18, 141–158.

National Conservation Strategy Secretariat (NCSS). 1993. National Conservation Strategy. National Policy on the Resources Base, its Utilization and Planning for Sustainability, Addis Ababa, Ethiopia.

Nelson, G.C., Geoghegan, J. 2002. Deforestation and land use change: sparse data environments. Agr. Econ. 27, 201–216.

Nguyen, T.T., Verdoodt, A., van Y, T., Delbecque, N., Tran, T.C., van Ranst, E. 2015. Design of a GIS and multi-criteria based land evaluation procedure for sustainable land-use planning at the regional level. Agric. Ecosyst. Environ. 200, 1–11.

Niedertscheider, M., Kuemmerle, T., Müller, D., Erb, K.-H. 2014. Exploring the effects of drastic institutional and socio-economic changes on land system dynamics in Germany between 1883 and 2007. Glob. Environ. Chang. 28, 98–108.

Njenga, M., Karanja, N., Munster, C., Iiyama, M., Neufeldt, H., Kithinji, J., Jamnadass, R. 2013. Charcoal production and strategies to enhance its sustainability in Kenya. Development in Practice 23, 359–371.

Nourqolipour, R., Mohamed Shariff, A.R.B., Balasundram, S.K., Ahmad, N.B., Sood, A.M., Buyong, T., Amiri, F. 2015. A GIS-based model to analyze the spatial and temporal development of oil palm land use in Kuala Langat district, Malaysia. Environ Earth Sci. 73, 1687–1700.

Obade, V., Lal, R. 2013. Assessing land cover and soil quality by remote sensing and geographical information systems (GIS). Catena. 104, 77–92.

Pacheco, F., Varandas, S., Sanches Fernandes, L., Valle Junior, R., 2014. Soil losses in rural watersheds with environmental land use conflicts. Sci. Total Environ. 485-486, 110–120.

Paré, S., Söderberg, U., Sandewall, M., Ouadba, J.M. 2008. Land use analysis from spatial and field data capture in southern Burkina Faso, West Africa. Agr. Ecosyst. Environ. 127, 277–285.

Paudel, S., Yuan, F. 2012. Assessing landscape changes and dynamics using patch analysis and GIS modeling. Int. J. Appl. Earth Obs. Geoinf. 16, 66–76.

Paz-Kagan, T., Shachak, M., Zaady, E., Karnieli, A. 2014. Evaluation of ecosystem responses to land-use change using soil quality and primary productivity in a semi-arid area, Israel. Agric. Ecosyst. Environ. 193, 9–24.

Petursson, J.G., Vedeld, P., Sassen, M. 2013. An institutional analysis of deforestation processes in protected areas: The case of the transboundary Mt. Elgon, Uganda and Kenya. For. Policy Econ. 26, 22–33.

Polasky, S., Nelson, E., Pennington, D., Johnson, K.A. 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.

Pontius, R., Cornell, J.D., Hall, C.A. 2001. Modeling the spatial pattern of land-use change with GEOMOD2: application and validation for Costa Rica. Agric. Ecosyst. Environ. 85, 191–203.

Prenzel, B. 2004. Remote sensing-based quantification of land-cover and land-use change for planning. Progr. Plan. 61, 281–299.

Pretty, J., Toulmin, C., Williams, S. 2011. Sustainable intensification in African agriculture. Int. J. Agric. Sustain. 9, 5–24.

Puyravaud, J.-P. 2003. Standardizing the calculation of the annual rate of deforestation. For. Ecol. Manage. 177, 593–596.

Qiang, Y., Lam, N.S.N. 2015. Modeling land use and land cover changes in a vulnerable coastal region using artificial neural networks and cellular automata. Environ. Monit. Assess. 187.

Rawlings, J.O., Pantula, S.G., Dickey, D.A. 1998. Applied regression analysis. A research tool. Springer, New York, 657pp.

Raynaut, C. 2001. Societies and nature in the Sahel: ecological diversity and social dynamics. Glob. Environ. Chang. 11, 9–18.

Reed, B.C., Brown, J.F., VanderZee, D., Loveland, T.R., Merchant, J.W., Ohlen, D.O. 1994. Measuring phenological variability from satellite imagery. J. Veg. Sci. 5, 703–714.

Reid, R.S., Kruska, R.L., Muthui, N., Taye, A., Wotton, S., Wilson, C.J., Mulatu, W. 2000. Landsc. Ecol. 15, 339–355.

Rembold, F., Carnicelli, S., Nori, M., Ferrari, G.A. 2000. Use of aerial photographs, Landsat TM imagery and multidisciplinary field survey for land-cover change analysis in the lakes region (Ethiopia). Int. J. Appl. Earth Obs. Geoinf. 2, 181–189.

Rindfuss, R.R., Walsh, S.J., Turner, B.L., Fox, J., Mishra, V. 2004. Developing a science of land change: Challenges and methodological issues. Proc. Natl. Acad. Sci. 101, 13976–13981.

Román-Cuesta, R.M., Carmona-Moreno, C., Lizcano, G., New, M., Silman, M., Knoke, T., Malhi, Y., Oliveras, I., Asbjornsen, H., Vuille, M. 2014. Synchronous fire activity in the tropical high Andes: an indication of regional climate forcing. Glob. Chang Biol. 20, 1929–1942.

Rutherford, G.N., Guisan, A., Zimmermann, N.E. 2007. Evaluating sampling strategies and logistic regression methods for modelling complex land cover changes. J. Appl. Ecol. 44, 414–424.

Sabatier, R., Doyen, L., Tichit, M. 2010. Modelling trade-offs between livestock grazing and wader conservation in a grassland agroecosystem. Ecol. Model. 221, 1292–1300.

Santos, R.G., Sturaro, J.R., Marques, M.L., Faria, T.T.d. 2015. GIS Applied to the Mapping of Land Use, Land Cover and Vulnerability in the Outcrop Zone of the Guarani Aquifer System. Procedia Earth Planet. Sci. 15, 553–559.

Schägner, J.P., Brander, L., Maes, J., Hartje, V. 2013. Mapping ecosystem services' values: Current practice and future prospects. Ecosyst. Serv. 4, 33–46.

Schaldach, R., Alcamo, J., Koch, J., Kölking, C., Lapola, D.M., Schüngel, J., Priess, J.A. 2011. An integrated approach to modelling land-use change on continental and global scales. Environ. Model. Softw. 26, 1041–1051.

Serneels, S., Lambin, E.F. 2001. Proximate causes of land-use change in Narok District, Kenya: a spatial statistical model. Agric. Ecosyst. Environ. 85, 65–81.

Shiferaw, A. 2011. Evaluating the Land use and Land Cover Dynamics in Borena Wereda of South Wollo Highlands, Ethiopia. J. Sustain. Dev. Africa 13, 87–107.

Shiferaw, B., Kassie, M., Jaleta, M., Yirga, C. 2014. Adoption of improved wheat varieties and impacts on household food security in Ethiopia. Food Policy 44, 272–284.

Singh, A. 1989. Review Article Digital change detection techniques using remotely-sensed data. Int. J. Remote Sens. 10, 989–1003.

Singh, P., Thakur, J.K., Kumar, S., Singh, U.C. 2012. Assessment of Land Use/Land Cover Using Geospatial Techniques in a Semi-arid Region of Madhya Pradesh, India. In: Thakur, J.K. (ed.), Geospatial techniques for managing environmental resources. Springer, Dordrecht, New York, pp. 152–163.

Singh, S. 2002. Balancing the approaches of environmental conservation by considering ecosystem services as well as biodiversity. Curr. Sci. 82, 1331–1335.

Sleeter, B.M., Sohl, T.L., Loveland, T.R., Auch, R.F., Acevedo, W., Drummond, M.A., Sayler, K.L., Stehman, S.V. 2013. Land-cover change in the conterminous United States from 1973 to 2000. Glob. Environ. Chang. 23, 733–748.

Soares-Filho, B.S., Nepstad, D.C., Curran, L.M., Cerqueira, G.C., Garcia, R.A., Ramos, C.A., Voll, E., McDonald, A., Lefebvre, P., Schlesinger, P. 2006. Modelling conservation in the Amazon basin. Nature 440, 520–523.

Spielman, D.J., Byerlee, D., Alemu, D., Kelemework, D. 2010. Policies to promote cereal intensification in Ethiopia: The search for appropriate public and private roles. Food Policy 35, 185–194.

Styers, D.M., Chappelka, A.H., Marzen, L.J., Somers, G.L. 2010. Developing a land-cover classification to select indicators of forest ecosystem health in a rapidly urbanizing landscape. Landsc.Urban Plan. 94, 158–165.

Sukara, E. 2014. Tropical Forest Biodiversity to Provide Food, Health and Energy Solution of the Rapid Growth of Modern Society. Procedia Environ. Sci. 20, 803–808.

Sun, X., Yue, T., Fan, Z. 2012. Scenarios of changes in the spatial pattern of land use in China. Procedia Environ. Sci. 13, 590–597.

Sylla, L., Xiong, D., Zhang, H., Bangoura, S. 2012. A GIS technology and method to assess environmental problems from land use/cover changes: Conakry, Coyah and Dubreka region case study. The Egyptian J. Remote Sens. Space Sci. 15, 31–38.

Taddese, G. 2001. Land Degradation: A Challenge to Ethiopia. Environ. Manage. 27, 815–824.

Taelman, S.E., Schaubroeck, T., Meester, S. de, Boone, L., Dewulf, J. 2016. Accounting for land use in life cycle assessment: The value of NPP as a proxy indicator to assess land use impacts on ecosystems. Sci. Total Environ. 550, 143–156.

Tefera, M.M. 2011. Land-Use/Land-Cover Dynamics in Nonno District, Central Ethiopia. J. Sustain. Develop. Africa 13, 123–141.

Tegene, B. 2002. Land-Cover/Land-Use Changes in the Derekolli Catchment of the South Welo Zone of Amhara Region, Ethiopia. East. Afr. Soc. Sci. Res. Rev. 8, 1–20.

Teixeira, A.M.G., Soares-Filho, B.S., Freitas, S.R., Metzger, J.P. 2009. Modeling landscape dynamics in an Atlantic Rainforest region: Implications for conservation. For. Ecol. Manage. 257, 1219–1230.

Teketay, D. 2000. Proceedings of the Round Table Conference on Integrated Forest Fire Management in Ethiopia. In: MOA, GTZ (eds.), Proceedings of the Round Table Conference on Integrated Forest Fire Management in Ethiopia. MoA, Addis Ababa.

Teketay, D., Granström, A. 1995. Soil seed banks in dry Afromontane forests of Ethiopia. J. Veg. Sci. 1995; 6: 777–786.

Teketay, D., Lemenih, M., Bekele, T., Yemshaw, Y., Feleke, S., Tadesse, W., Moges, Y., Hunde, T., Nigussie, D. 2010. Forest Resources and Challenges of Sustainable Forest Management and Conservation in Ethiopia. In: Bongers, F., Tennigkeit, T. (eds.), Degraded forests in Eastern Africa. Management and restoration. Earthscan, London, Washington, DC, pp. 19–64.

Tekle, K., Hedlund, L. 2000. Land Cover Changes Between 1958 and 1986 in Kalu District, Southern Wello, Ethiopia. Mt. Res. Dev. 20, 42–51.

Temesgen, H., Nyssen, J., Zenebe, A., Haregeweyn, N., Kindu, M., Lemenih, M., Haile, M. 2013. Ecological succession and land use changes in a lake retreat area (Main Ethiopian Rift Valley). J. Arid Environ. 91, 53–60.

Tesfaye, G., Teketay, D., Fetene, M., Beck, E. 2010. Regeneration of seven indigenous tree species in a dry Afromontane forest, southern Ethiopia. Flora Mor. Dist. Fun. Ecol. Plants 205, 135–143.

Tesfaye, G., Wolff, M. 2014. The state of inland fisheries in Ethiopia: a synopsis with updated estimates of potential yield. Ecohydrol. Hydrobiol. 14, 200–219.

Thiombiano, L., Tourino-Soto, I. 2007. Status and Trends in Land Degradation in Africa. In: Sivakumar, M.V.K. (ed.), Climate and land degradation. With 61 tables. Springer, Berlin, pp. 39–53.

Thomlinson, J.R., Bolstad, P.V., Cohen, W.B. 1999. Coordinating Methodologies for Scaling Land cover Classifications from Site-Specific to Global. Remote Sen. Environ. 70, 16–28.

Tian, H., Banger, K., Bo, T., Dadhwal, V.K. 2014. History of land use in India during 1880–2010: Large-scale land transformations reconstructed from satellite data and historical archives. Glob. Planet. Chang. 121, 78–88.

Tian, J., Chen, D. 2007. Optimization in multi-scale segmentation of high-resolution satellite images for artificial feature recognition. Int. J. Remote Sens. 28, 4625–4644.

Tianhong, L., Wenkai, L., Zhenghan, Q. 2010. Variations in ecosystem service value in response to land use changes in Shenzhen. Ecol. Econ. 69, 1427–1435.

Tigabu, M., Lemenih, M., Negash, M., Teketay, D. 2014. Rehabilitation of degraded forest and woodland ecosystems in Ethiopia for sustenance of livelihoods and ecosystem services. In: Katila, P., Galloway, G., Jong, W. de, Pacheco, P., Mery, G. (eds.), Forests under pressure. Local responses to global issues. International Union of Forest Research Organizations (IUFRO), Vienna, pp. 299–313.

Toman, M. A. 1998. Why not to calculate the value of the worldvs ecosystem services and natural capital Ecol. Econ. 25, 57–60.

Troy, A., Wilson, M.A. 2006. Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer. Ecol. Econ. 60, 435–449.

Tsegaye, D., Moe, S.R., Vedeld, P., Aynekulu, E. 2010. Land-use/cover dynamics in Northern Afar rangelands, Ethiopia. Agric. Ecosyst. Environ. 139, 174–180.

Turner, I.B. 1997. The Sustainability Principle in Global Agendas: Implications for Understanding Land-Use/Cover Change. Geogr. J. 163, 133.

Turner, R., Paavola, J., Cooper, P., Farber, S., Jessamy, V., Georgiou, S. 2003. Valuing nature: lessons learned and future research directions. Ecol. Econ. 46, 493–510.

United Nations, Department of Economic and Social Affairs (UN DESA). 2013. World Population Prospects: The 2012 Revision, Highlights and Advance Tables. United Nations, New York, 118 pp.

Valle Junior RF, Varandas SGP, Sanches Fernandes LF, Pacheco FAL., 2014a. Environmental land use conflicts: A threat to soil conservation. Land Use Policy 41:172–85.

Valle Junior, R., Varandas, S., Sanches Fernandes, L., Pacheco, F., 2014b. Groundwater quality in rural watersheds with environmental land use conflicts. ci. Total Environ. 493, 812–827.

Valle Junior, R., Varandas, S.G., Pacheco, F.A., Pereira, V.R., Santos, C.F., Cortes, R.M., Sanches Fernandes, L.F., 2015. Impacts of land use conflicts on riverine ecosystems. Land Use Policy 43, 48–62.

van der Ploeg, S., de Groot, D. 2010. The TEEB Valuation Database – a searchable database of 1310 estimates of monetary values of ecosystem services, Wageningen, The Netherlands.

Verburg, P.H., Schot, P.P., Dijst, M.J., Veldkamp, A. 2004. Land use change modelling: current practice and research priorities. Geoiournal 61, 309–324.

Verburg, P.H., Soepboer, W., Veldkamp, A., Limpiada, R., Espaldon, V., Mastura, S.S. 2002. Modeling the Spatial Dynamics of Regional Land Use: The CLUE-S Model. Environ. Manage. 30, 391–405.

Wale, E. 2008. A study on financial opportunity costs of growing local varieties of sorghum in Ethiopia: Implications for on-farm conservation policy. Ecol. Econ. 64, 603–610.

Wang, E., Bell, M., Luo, Z., Moody, P., Probert, M.E. 2014. Modelling crop response to phosphorus inputs and phosphorus use efficiency in a crop rotation. Field Crops Res. 155, 120–132.

Wang, Y., Gao, J., Wang, J., Qiu, J., Bond-Lamberty, B. 2014. Value Assessment of Ecosystem Services in Nature Reserves in Ningxia, China: A Response to Ecological Restoration. PLoS ONE 9, e89174.

Wang, Z., Zhang, B., Zhang, S., Li, X., Liu, D., Song, K., Li, J., Li, F., Duan, H. 2006. Changes of Land Use and of Ecosystem Service Values in Sanjiang Plain, Northeast China. Environ. Monit. Assess. 112, 69–91.

Wardell, D., Reenberg, A., Tøttrup, C. 2003. Historical footprints in contemporary land use systems: forest cover changes in savannah woodlands in the Sudano-Sahelian zone. Glob .Environ. Chang. 13, 235–254.

Weng, Q. 2002. Land use change analysis in the Zhujiang Delta of China using satellite remote sensing, GIS and stochastic modelling. J. Environ. Manage. 64, 273–284.

Westman, W.E. 1977. How Much Are Nature's Services Worth? Science 197, 960–964.

Whiteside, T.G., Boggs, G.S., Maier, S.W. 2011. Comparing object-based and pixel-based classifications for mapping savannas. Int. J. Appl. Earth Obs. Geoinf. 13, 884–893.

Wondie, M., Schneider, W., Melesse, A.M., Teketay, D. 2011. Spatial and Temporal Land Cover Changes in the Simen Mountains National Park, a World Heritage Site in Northwestern Ethiopia. Remote Sens. 3, 752–766.

Wood, E., Tappan, G., Hadj, A. 2004. Understanding the drivers of agricultural land use change in south-central Senegal. J. Arid Environ. 59, 565–582.

Woodhouse, P. 2012. New investment, old challenges. Land deals and the water constraint in African agriculture. J. Peasant Stud. 39, 777–794.

World Summit on Food Security (WSFS). 2009. Declaration of the World Summit on Food Security, Food and Agriculture Organization of the United Nation.

Wu, Q., Li, H.-q., Wang, R.-s., Paulussen, J., He, Y., Wang, M., Wang, B.-h., Wang, Z. 2006. Monitoring and predicting land use change in Beijing using remote sensing and GIS. Landsc. Urban Plan. 78, 322–333.

Xiao, J., Shen, Y., Ge, J., Tateishi, R., Tang, C., Liang, Y., Huang, Z. 2006. Evaluating urban expansion and land use change in Shijiazhuang, China, by using GIS and remote sensing. Landsc. Urban Plan. 75, 69–80.

Xiuqin, W., Yunlong, C., Tao, Z. 2011. Effects of Land Use/Land Cover Changes on Rocky Desertification-A Case Study of a Small Karst Catchment in Southwestern China. Energy Procedia 5, 1–5.

Yan, G., Mas, J.-F., Maathuis, B.H.P., Xiangmin, Z., van Dijk, P.M. 2006. Comparison of pixel-based and object-oriented image classification approaches—a case study in a coal fire area, Wuda, Inner Mongolia, China. Int. J. Remote Sens. 27, 4039–4055.

Yang, Y., Feng, Z., Huang, H.Q., Lin, Y. 2008. Climate-induced changes in crop water balance during 1960–2001 in Northwest China. Agric. Ecosyst. Environ. 127, 107–118.

Yoshida, A., Chanhda, H., Ye, Y.-M., Liang, Y.-R., 2010. Ecosystem service values and land use change in the opium poppy cultivation region in Northern Part of Lao PDR. Acta Ecol. Sin. 30, 56–61.

Zaimes, G.N., Schultz, R.C. 2015. Riparian land-use impacts on bank erosion and deposition of an incised stream in north-central Iowa, USA. Catena 125, 61–73.

Zeleke, G., Hurni, H. 2001. Implications of Land Use and Land Cover Dynamics for Mountain Resource Degradation in the Northwestern Ethiopian Highlands. Mt. Res. Dev. 21, 184–191.

Zhang, Z., Gao, J., Fan, X., Lan, Y., Zhao, M., 2017. Response of ecosystem services to socioeconomic development in the Yangtze River Basin, China. Ecol. Indic. 72, 481–493.

Zhao, B., Kreuter, U., Li, B., Ma, Z., Chen, J., Nakagoshi, N. 2004. An ecosystem service value assessment of land-use change on Chongming Island, China. Land Use Policy 21, 139–148.

Appendix: Publications

Publication I

Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2013: Land Use/Land Cover Change Analysis Using Object-Based Classification Approach in Munessa-Shashemene Landscape of the Ethiopian Highlands. Remote Sensing 5 (5): 2411-2435. doi:10.3390/rs5052411

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Article

Land Use/Land Cover Change Analysis Using Object-Based Classification Approach in Munessa-Shashemene Landscape of the Ethiopian Highlands

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Abstract: The objective of this study was to analyze land use/land cover (LULC) changes in the landscape of Munessa-Shashemene area of the Ethiopian highlands over a period of 39 years (1973–2012). Satellite images of Landsat MSS (1973), TM (1986), ETM+ (2000), and RapidEye (2012) were used. All images were classified using object-based image classification technique. Accuracy assessments were conducted for each reference year. Change analysis was carried out using post classification comparison in GIS. Nine LULCs were successfully captured with overall accuracies ranging from 85.7% to 93.2% and Kappa statistic of 0.822 to 0.924. The classification result revealed that grasslands (42.3%), natural forests (21%), and woodlands (11.4%) were dominant LULC types in 1973. In 2012, croplands (48.5%) were the major LULC types followed by others. The change result shows that a rapid reduction in woodland cover of 81.8%, 52.3%, and 36.1% occurred between the first (1973–1986), second (1986–2000), and third (2000–2012) study periods, respectively. Similarly, natural forests cover decreased by 26.1% during the first, 21.1% during the second, and 24.4% during the third periods. Grasslands also declined by 11.9, 17.5, and 21.1% during the three periods, respectively. On the contrary, croplands increased in all three periods by 131, 31.5, and 22.7%, respectively. Analysis of the 39-year change matrix revealed that about 60% of the land showed changes in LULC. Changes were also common along the slope gradient and agro-ecological zones with varying proportions. Further study is suggested to investigate detailed drivers and consequences of changes.

Keywords: Landsat; RapidEye; accuracy assessment; remote sensing; GIS; image; Ethiopia

1. Introduction

Land use/land cover (LULC) changes influence climate and weather conditions from local to global scales [1]. They can have such impacts by affecting the composition of the atmosphere and the exchange of energy between continents and the atmosphere, which can lead to global warming [2]. Changes in LULC can also affect biological diversity, contribute to forest fragmentation, lead to soil erosion, alter ecosystem services, disrupt socio-cultural practices, and increase natural disasters, such as flooding [3,4]. This calls for global attention for continuous monitoring of the changes. Up-to-date datasets on LULC change provide critical inputs to evaluate complex causes and responses in order to project future trends better, ranging from local, regional, to global scales [5,6]. They are also prerequisites for making development plans [7,8]. However, the magnitude of LULC change differs with the time period being examined [9], geographical location [10], slope gradient, and elevation range [11,12].

With an area of 1,130,000 km^2, and as one of the most populous countries in Africa, Ethiopia is experiencing huge LULC dynamics from natural vegetation to farming practices and human settlement [13,14]. The problem of land cover dynamics is more severe in the highlands, which account nearly 44% of the country's landmass and have been cultivated for millennia [15,16]. Like other parts of the world, the use and management of natural resources, and returning the vast degraded landscapes to protective and/or productive systems, have substantial importance to attain the goal of sustainable development in Ethiopia [17]. This, in turn, requires an understanding of the dynamics in time and space of these resources. In this regard, the importance of spatial data monitoring and evaluation for proper management of natural resources is critical. Some studies have been conducted to estimate and monitor LULC changes in different parts of the Ethiopian highlands [18–26]. These reports have shown heterogeneity in direction, pattern, type, and/or magnitude of LULC changes in the country. For instance, Zeleke and Hurni [21] reported a sharp decrease of forest cover while Bewket [22] found the opposite, *i.e.*, an increasing trend. In terms of magnitude for changes, Zeleke and Hurni [21] reported an increase in cultivated lands by 38% in 38 years (1957–1995). On the other hand, Tegene [23] reported an increase in croplands only by 5.5% in 43 years (1957–2000). Consequently, making generalizations of results to other areas of the same physical setting might lead to erroneous conclusions. In addition, except the reports by Zeleke and Hurni [21] and Tegene [23] about changes in relation to slope gradient, no studies have been conducted in comprehensive approach to systematically analyze changes within the study area along slope gradient and agro-ecological zone, which is usually divided based on elevation range [27]. Therefore, it can be concluded that, until now, few studies have undertaken an integrated analyses on LULC change in the Ethiopian Highlands.

Results of various studies have demonstrated the need for a study focusing on location specific LULC dynamics for sustainable management and decision-making processes related to the use and conservation of natural resources [22,26,28,29]. Two ways of capturing LULC dynamics are available:

conventional ground- and remote sensing-based methods. The ground method is labor intensive, time consuming, and difficult for capturing data from inaccessible areas with ragged topographies like the case of most Ethiopian landscape. On the contrary, remote sensing is considered the most efficient technology to handle these problems since it can explicitly reveal spatial patterns of land cover change over a large geographic area in a regular and consistent way [30,31]. Remote sensing data of the earth's surface could be made readily available in digital format [32]. These advantages have attracted great interest in the scientific community. Moreover, the rich archive and spectral resolution of satellite images are the most important reasons for their use [30,33]. Thus, change detection has become a major application of remotely-sensed data because of repetitive coverage at short time intervals, which is useful for tracking changes in LULC over longer periods of time and at more varied temporal scales than what is typically done with field experiments or ground inventory [9,34]. Various techniques are available to extract meaningful information of LULCs from remotely captured datasets.

Recently, object-based image analysis has been applied more frequently for remote sensing image classification than pixel-based analysis [35,36]. Pixel-based methods classify individual pixels mainly using spectral patterns. The use of spatial or contextual information from neighborhood pixels remains a critical drawback to pixel-based image processing [37]. On the other hand, object-based methods allow integration of different object features, such as spectral values, shape, and texture [38–40]. One of its strength is the ability to combine spectral information and spatial information for extracting target objects [36,38].

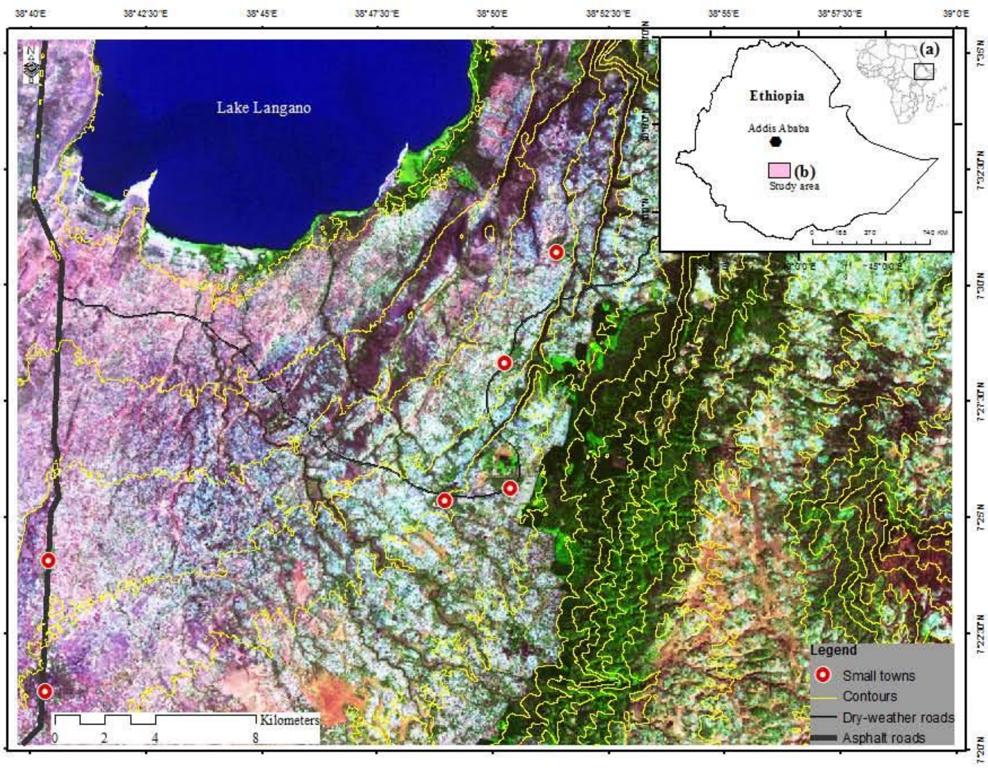
However, accuracies of object-based approach differ depending on the nature of landscape and type of images used for analysis [41]. The benefit of improving accuracies of image classifications using object-based approach is not tested up to now in the Ethiopian landscape during LULC change studies. Studies conducted so far in the country were pixel-based and their overall accuracies from recent reports were not more than 88% [25,26]. On the other hand, apart from the need for location specific LULC change study justified above in a landscape with diverse features like the Ethiopian highlands, the necessity for improved classification accuracies for LULC change studies in the country have been discussed by many researchers [23,25,26,42,43]. Thus, in this study, we classified the land use/land cover with the highest possible accuracy and evaluated changes over a period of 39 years (1973–2012) in a landscape of Munessa-Shashemene area, one of the typical highlands in Ethiopia. We also explored distribution and changes in LULC along the slope gradient and Agro-ecological zones of the study landscape.

2. Materials and Methods

2.1. Study Area

The study was conducted in the landscape of Munessa-Shashemene area, which is a typical highland found in Munessa and Arsi-Negele Districts. The area lies within 7°20'01.23" and 7°35'13.3"N, and 38°39'43.3" and 38°59'57.31"E at about 200 km south of Addis Ababa (Figure 1). It covers about 1,091 km^2 and lies between the altitudes of 1,500 m above sea level at the Central Rift Valley lakes and over 3,400 m at the Arsi-Bale massif. The rainfall has bimodal distribution. The short and main rainy seasons occur from March–May and July–September, respectively. Meteorological station records show that annual rainfall is about 1,200 mm at Degaga town (2,000 m), which is found in the study area. Mean annual temperature is 15 °C. The soils of the area are rich in clay and classified as Mazic Vertisol in the lower altitude (1,500 m) and Humic Umbrisol at about 3,000 m [44]. It is a diverse landscape with both flat and sloped areas. Crop cultivation is common in all altitudinal ranges with various proportions. Apart from croplands, the study landscape comprises mosaics of LULC types, mainly natural forests, plantation forests, woodlands, settlements and water bodies. The natural forest in the area belongs to a tropical dry Afromontane forest [45]. The plantation forests are composed of exotic species, mainly *Cupresses lusitanica* Miller, *Pinus patula* Schlechtendal & Chamisso and *Eucalyptus* spp. Woodlands are dominated by *Acacia* spp and found in the lower part of the study landscape.

Figure 1. Location of the study area: (**a**) African context, (**b**) landscape level study location in reference to Ethiopia. Background is Landsat 7 Enhanced Thematic Mapper (ETM+) image of the year 2000 with RGB (Red, Green, Blue): Band 5, band 4 and 3. Topographic counters with 100 m were developed from Aster Digital Elevation Model (DEM).



2.2. Data Used

Datasets from various sources were used in this study (Table 1). Landsat and RapidEye imagery were the main data for classification and change analysis. The Landsat imagery data include Landsat MSS, Landsat Thematic Mapper (TM), and Enhanced Thematic Mapper (ETM+) scenes of the year 1973, 1986 and 2000, respectively. These datasets were acquired from the National Aeronautics and

Space Administration (NASA) through their EOS Data Gateway Database. The RapidEye imagery data were used for the study year 2012. The images were obtained from RapidEye Science Archive (RESA) supported by German Aerospace Center (DLR). They were ortho-rectified level 3A and geometrically corrected. Both Landsat and RapidEye imageries were acquired in the same season. Images of the same season were selected to reduce the effect of seasonal discrepancies on the classification result [46].

Table 1. Summary of spatial datasets used in this study.

Dataset Type	Acquisition Date (s)	Pixel Resolution/Scale
Satellite data		
Landsat MSS	30.1.1973	60 m
Landsat TM 5	21.1.1986	30 m
Landsat ETM+	5.2.2000	30 m
RapidEye	1.2.2012	5 m
Aster DEM		30 m
Aerial photo		
Black and white	1.2.1972	1:50000
Black and white	1.2.1986	1:20000
Ancillary data		
Topo map		1:50000
Field data	November 2011–January 2012	
Roads and towns		
Kebele boundary		
Wereda boundary		

A 30 m Digital Elevation Model (DEM), based on Aster imagery, was also employed in order to study the relationship between LULC types with that of the slope gradient and agro-ecological zones of the study landscape. In addition, ancillary data were also utilized during analysis, including topographic maps, field data, thematic layers (roads and towns), Kebele and Wereda boundaries. All data were projected to the Universal Transverse Mercator (UTM) projection system zone 37N and datum of World Geodetic System 84 (WGS84), ensuring consistency between datasets during analysis. The pre-processing was made using ArcGIS 10 software. Fieldwork was conducted between November 2011 and January 2012 using draft classified maps derived from satellite images with reference years, aerial photos and topographic maps as guides. GPS coordinates of target LULC types were collected, and information regarding each site was noted. Thematic layers of towns were used to facilitate the classification process. Roads with other datasets were utilized for the study site map preparation.

2.3. Methodology

2.3.1. Image Segmentation

Object-based image analysis requires the creation of objects or separated regions in an image. One established way to do so is image segmentation. Depending on its application, different approaches exist for image segmentation ranging from very simple to highly sophisticated algorithm [40].

We used the sophisticated segmentation algorithm, known as multi-resolution segmentation (MS), which is based on the Fractal Net Evolution Approach (FNEA) [38] and available in eCognition Developer 8.0 software. The MS algorithm is bottom-up region merging technique starting with a single image object of one pixel and repeatedly merges them in several loops in pairs to larger units. The MS algorithm is also an optimization procedure that minimizes the average heterogeneity for a given number of objects and maximizes their homogeneity based on defined parameters. These parameters, namely scale (Sc), shape (Sh), and compactness (Cm), are defined through trial and error to successfully segment objects in an image [40,47–49]. We used scale parameters ranging from 8 to 500 with three different levels depending on the type of images used for the analysis (Table 2). The images were segmented in to three levels to facilitate the object-based classification depending on the nature of LULC classes to be detected. For instance, level 1 was to handle those big size classes like water bodies, whereas level 3 was for small size classes like tree patches. Segmentation outputs were visually checked in relation to target class (e.g., forest area or cropland) to evaluate which parameter combinations best captured the objects of interest.

Table 2. Parameters used for different images in each segmentation level.

Data Type	Resolution	Parameters Used in Different Segmentation Levels								
		Parameters for Level 1			Parameters for Level 2			Parameters for Level 3		
		Sc [a]	Sh [b]	Cm [c]	Sc	Sh	Cm	Sc	Sh	Cm
Landsat MSS	60 m	40	0.2	0.8	15	0.2	0.8	8	0.2	0.8
Landsat TM 5	30 m	50	0.1	0.5	20	0.1	0.5	10	0.1	0.5
Landsat ETM+	30 m	50	0.1	0.5	20	0.1	0.5	10	0.1	0.5
RapidEye	5 m	500	0.2	0.7	200	0.2	0.7	50	0.2	0.7

[a] Scale (Sc) parameter identifies the highest heterogeneity allowed for the objects; [b] Shape (Sh) parameter balances spectral homogeneity versus shape of objects on segmentation outcome; [c] Compactness (Cm) parameter determines image objects based on their relative shape.

2.3.2. Object-Based Classification

We used various sources, including field survey, ancillary data and existing Afri-cover classification approaches to set and implement our object-based classification schemes. Nine LULC classes were considered for this purpose (Table 3). Considering the power of object-based methods, an attempt has been made to separate forest types, e.g., natural forests and woodlands, in our classification schemes. This is owing to their differences in providing services and goods, and the underlying pressures towards the resources. Classifying them separately can also facilitate conservation, utilization, and management approaches.

Using identified target LULC classes, object-based classification was applied to a segmented image in order to assign a class to each of the segments. Object-based image analysis attempts to assign objects that are generated through image segmentations into a specific class of interest. We used eCognition to perform an object-based image classification [40]. There are two approaches in eCognition to assign classes to segmented objects, which are fuzzy membership functions and the nearest neighbor (NN) classifier. The membership function classifier uses the user's expert knowledge to define rules and constraints in the membership function to control the classification procedure. The membership function describes intervals of object features that determine whether the objects belong to a particular class or not. An object feature can be the spectral value, texture, size, shape, and context of that image object to surrounding image objects. On the other hand, NN classifier uses a defined feature space, e.g., using original bands or customized bands, and a set of samples that represent different classes in order to assign class values to segmented objects. The procedure consists of teaching the system by giving certain image objects as samples and classifying image objects in the image object domain based on their nearest sample neighbors. Employing NN classifier is advantageous when using spectrally similar classes that are not well separated using a few features or just one feature [40]. Whenever applicable, we used both approaches during the classification process.

Table 3. Description of land use/land covers classes used for change study from 1973 to 2012.

LULC Types	General Description
Bare lands	Areas of land that already gets bad either due to erosion or misuses especially over grazing and crop cultivation.
Grasslands	All areas covered with natural grass and small shrubs dominated by grass.
Water	Permanent lakes and other intermittent ponds.
Settlements	Build-ups (houses) in both urban and rural parts.
Croplands	Areas of land prepared for growing agricultural crops. This category includes areas currently under crop, and land under preparation.
Tree patches	Areas covered with scattered trees, bushes and shrubs along the study landscape. Small patches of forests (<0.5 ha) are also included in this category.
Plantation forests	Areas covered by man-made trees with minimum size of 0.5 ha.
Natural forests	Areas dominated by natural high forests, which are coniferous or deciduous.
Woodlands	Forests found below 1900 m a.s.l. [51]. Mainly dominated by *Acacia* spp.

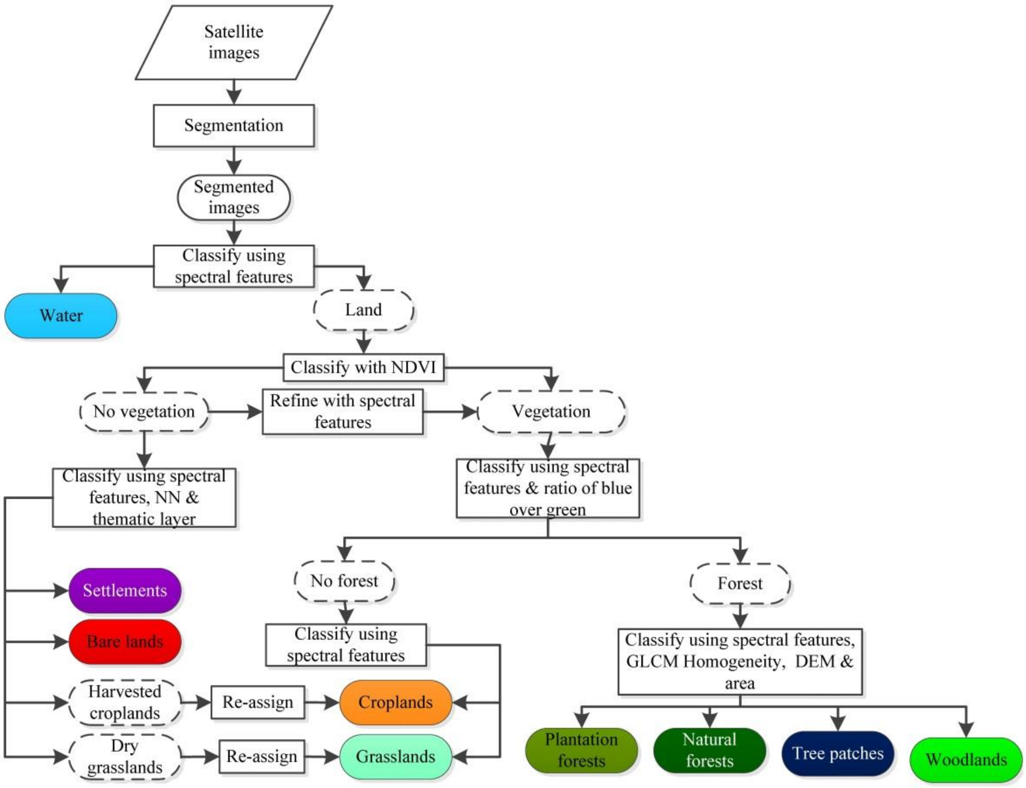
A hierarchical scheme of three levels was implemented during object-based classification using eCognition 8.0 software. The classification was applied using a ‘‘top-down’’ approach. That is, the classification started from very general classes (level 1), which were further subdivided into more specific classes (level 2 and 3). We first broadly classified the whole study landscape into water and land classes by using the spectral features from the mean value of objects in near infrared band. The second and third levels were used to extract the remaining target LULC types from the class land (Figure 2).

The classification of target classes was achieved by using mainly thresholds of mean and/or standard deviation of spectral features (original bands of blue, green, red, red edge, and near infrared), customized bands (ratio of blue over green), thematic layers, DEM values, texture value of grey-level co-occurrence matrix (GLCM) homogeneity, and normalized difference vegetation index (NDVI). The NDVI was calculated using the following equation:

$$\mathrm{NDVI} = \frac{\mathrm{NIR - RED}}{\mathrm{NIR + RED}} \tag{1}$$

where NIR and RED are reflectance in the near infrared and the red bands, respectively. The NDVI values have been typically used to map spatial distributions of vegetation [50].

Figure 2. Classification scheme used in eCognition for LULC classification in the study landscape. Dash line boxes are intermediate/temporal classes and colored boxes are final LULC classes.



In this study, NDVI values were also used to further classify the class land into vegetation and no-vegetation classes. As the images were taken during the dry season, some dried vegetation with low NDVI was classified as no-vegetation class. We used our expert knowledge and developed a rule set using red and red edge mean values to refine such classes from no-vegetation to vegetation classes. The vegetation class was again subdivided into forest and no-forest classes. The mean value of objects in red band and a value from blue over green ratio were used to differentiate these two categories, e.g., higher values are forests where as lower values are no-forest classes. The forest class was again classified to achieve the final target LULC types, namely plantation forests, tree patches, natural forests, and woodlands. The standard deviation value of objects in red band, texture value of GLCM homogeneity, DEM, and size (area) of the class objects were utilized to separate these four classes. The DEM was particularly used to separate woodlands from natural forests. Woodlands are usually found below 1,900 m [51]. Plantation forests have fine texture than natural forests. As a result, the texture values of GLCM homogeneity with other values were used to differentiate these two classes. On the other hand, the mean value of NIR was utilized to separate grasslands and croplands from the class no-forest. Based on the training samples, the NN classifier was also employed to further classify the no-vegetation class into settlements, bare lands, harvested croplands, and dry grasslands. Again, associated to the dry season images acquisition, there were areas that were signed as bare lands although they were dry grasslands and harvested croplands. To avoid such confusion, the two classes (harvested croplands and dry grasslands) were temporally created under no-vegetation class. Mean values of blue, red, and NIR bands were used to define the feature space during the classification process using NN classifier. Additionally, mean value of red and blue bands and other features, such as thematic layer of towns in the study landscape, were also used to define rules and constraints in the membership function to refine the classification process of these particular classes. At the end, the harvested croplands and dry grasslands were re-assigned to the class croplands and grasslands, respectively, using the 'assign class' function available in eCognition Developer 8.0 software.

2.3.3. Accuracy Assessment

A classification is not complete until its accuracy is assessed [30]. Accuracy assessment determines the quality of the map extracted from remotely sensed data [52,53]. An error matrix or confusion matrix is a common practice employed for assessment of classification accuracy. The matrix compares information obtained by reference sites to that provided by classified image for a number of sample areas. Accordingly, overall accuracy, producer's and user's accuracies, and Kappa statistic were calculated from the error matrix [53].

The accuracy assessments were performed for classified images of 1973, 1986, 2000, and 2012. A minimum of about 40 random points were generated per class using stratified random sampling approach for efficient accuracy assessment [53]. The corresponding reference class for each LULC type was collected from different data sources, including data from field visits, historical black and white aerial photos, topographic maps, and raw images. Raw images were used for those visually visible classes, e.g., forests and water bodies [53]. Historical black and white aerial photos (1973 and 1986) were utilized to collect reference samples for the 1973 and 1986 classified images while field visits data were mainly used for the 2012 classified image. Reference points for the 2000 classified image were collected through visual interpretation of the raw Landsat TM 2000 image. This was supplemented by field visits and discussion with elders in the study landscape that made it possible to establish reference points of confusing classes, e.g., croplands and grasslands.

2.3.4. Land Use/Land Cover Change Analysis

Change analysis was conducted using post-classification image comparison technique [54]. Post-classification change analysis was selected in order to minimize possible effects of atmospheric variations and sensor differences (Lu *et al.* [46]). Classification with high accuracy is a prerequisite for effective change detection using post-classification technique [55]. Images of different reference years were first independently classified. Classified images with highest accuracy were used in the change detection process.

The classified images were compared in three periods, *i.e.*, 1973–1986, 1986–2000, and 2000–2012. Change statistics were computed by comparing image values of one data set with the corresponding value of the second data set in each period. This results in a summary table of the overall changes per class. The values were presented in terms of hectares and percentages. The percentage LULC changes were calculated using the following equation:

$$\text{Percentage LULC change} = \left(\frac{\text{Area}_{\text{final year}} - \text{Area}_{\text{initial year}}}{\text{Area}_{\text{initial year}}}\right) \times 100 \quad (2)$$

where Area is extent of each LULC type. Positive values suggest an increase whereas negative values imply a decrease in extent. LULC conversion matrix between 1973 and 2012 was generated using ArcGIS 10 software and compiled in a matrix table, and the values were presented in terms of hectares.

Slope gradients and agro-ecological zones (AEZs) are conditional factors for LULC distributions and changes [27,56]. The slope gradient was calculated from Aster DEM in ArcGIS environment. It was reclassified into four slope categories, *i.e.*, gentle slopes (0%–5%), moderate slopes (5%–15%), steep slopes (15%–30%), and very steep slopes (>30%) [57]. AEZs of the study landscape were developed using elevation range of Aster DEM, based on categories of Ministry of Agriculture (MOA, [58]) and Hurni [27]. The categories include sub-humid highlands (locally known as "Weyna Dega") from 1,500 to 2,300 m, humid highlands ("Dega") from 2,300 to 3,200 m and cold highlands "Wurch") above 3,200 m. By overlaying the classified maps of each reference year (1973, 1986, 2000, and 2012) on to the slope and AEZ map, thematic information showing relationship between LULC distribution and changes in each category was extracted in ArcGIS.

3. Results and Discussion

3.1. Results

3.1.1. Classification Accuracy

The overall accuracies for the four reference years ranged from 85.7% to 93.2% with the Kappa statistic ranging from 0.822 to 0.924 (Table 4). The Kappa results show a high level of agreement for each of the four classified images. User's and producer's accuracies of individual classes of the four classified maps ranged from 72.1% (bare lands in 1973) to 100% (water in 2012), and 75.7% (grassland in 1973) to 100% (water in 2012), respectively. Our producer and user accuracies of the Landsat MSS classified image of 1973 for bare lands, croplands, grasslands, and settlements (user's accuracy) were below 85%. There were samples, which were identified as bare lands that actually belonged to croplands, grasslands, and settlements (Table 4(a)). On the other hand, the natural forests, tree patches, woodlands, and water classes achieved producer and user accuracies of above 90%. Confusions in these classes were low. For instance, in the second column of Table 4(a), out of the total 61 randomly generated reference samples for natural forests, 59 were correctly classified as natural forests, while only one sample was excluded or misclassified as cropland and the other one sample as tree patch. Similarly, in the second row of Table 4(a), three samples of grasslands and two samples of tree patches were included or misclassified as natural forests. The individual accuracies of these classes were more than 93% except for tree patches (producer's accuracy in 1986) for classified images of both 1986 and 2000. Again, the individual accuracies of these categories reached above 96% for 2012 classified image (Table 4(d)).

Table 4. Error matrix of classification accuracies for (**a**) 1973, (**b**) 1986, (**c**) 2000 and (**d**) 2012.

Classified Data	(a) Reference Data								Total	UA(%)
	Bare Lands	**Natural Forests**	**Crop Lands**	**Grass Lands**	**Settlements**	**Tree Patches**	**Wood Lands**	**Water**		
Bare lands	**31**	0	8	2	2	0	0	0	43	72.1
Natural forests	0	**59**	0	3	0	2	0	0	64	92.2
Croplands	0	1	**52**	5	0	1	4	0	63	82.5
Grasslands	5	0	2	**56**	1	2	2	0	68	82.4
Settlements	1	0	5	4	**39**	3	0	0	52	75.0
Tree patches	0	1	0	1	1	**53**	2	0	58	91.4
Woodlands	0	0	0	2	0	3	**56**	1	62	90.3
Water	0	0	0	1	0	0	2	**57**	60	95.0
Total	37	61	67	74	43	64	66	58	470	
PA(%)	83.8	96.7	77.6	75.7	90.7	82.8	84.8	98.3		

Overall accuracy = 85.7%; Kappa statistic = 0.822.

Classified Data	(b) Reference Data									Total	UA (%)
	Bare Lands	**Natural Forests**	**Plantation Forests**	**Crop Lands**	**Grass Lands**	**Settlements**	**Tree Patches**	**Wood Lands**	**Water**		
Bare lands	**42**	0	0	6	3	0	0	0	0	51	82.4
Natural forests	0	**59**	3	0	0	0	0	0	0	62	95.2
Plantation forests	0	2	**52**	0	0	0	0	0	0	54	96.3
Croplands	5	0	0	**54**	3	3	0	0	0	65	83.1
Grasslands	1	1	0	1	**57**	1	2	1	0	64	89.1
Settlements	0	0	0	0	4	**47**	4	0	0	55	85.5
Tree patches	0	0	0	0	4	0	**57**	0	0	61	93.4
Woodlands	0	0	0	1	2	0	0	**48**	0	51	94.1
Water	0	0	0	0	1	0	0	0	**53**	54	98.1
Total	48	62	55	62	74	51	63	49	53	517	
PA(%)	87.5	95.2	94.5	87.1	77	92.2	90.5	98	100		

Overall accuracy = 90.7%; Kappa statistic = 0.895.

Classified Data	(c) Reference Data									Total	UA (%)
	Bare Lands	**Natural Forests**	**Plantation Forests**	**Crop Lands**	**Grass Lands**	**Settlements**	**Tree Patches**	**Wood Lands**	**Water**		
Bare lands	**43**	0	0	6	5	1	0	0	0	55	78.2
Natural forests	0	**57**	2	0	0	0	0	0	0	59	96.6
Plantation forests	0	1	**55**		0	0	0	0	0	56	98.2
Croplands	3	0	0	**59**	4	2	0	0	0	68	86.8
Grasslands	2	0	0	2	**54**	1	1	1	0	61	88.5
Settlements	0	0	0	1	3	**49**	3	0	0	56	87.5
Tree patches	0	0	0	0	2	2	**58**	0	0	62	93.5
Woodlands	0	0	0	1	2	0	0	**45**	0	48	93.8
Water	0	0	0	0	1	0	0	0	**51**	52	98.1
Total	48	58	57	69	71	55	62	46	51	517	
PA(%)	90.4	98.3	96.5	90.8	76.1	89.1	93.5	97.8	100		

Overall accuracy = 91.9; Kappa statistic = 0.908.

Table 4. *Cont.*

Classified Data	(d) Reference Data									Total	UA (%)
	Bare Lands	Natural Forests	Plantation Forests	Crop Lands	Grass Lands	Settlements	Tree Patches	Wood Lands	Water		
Bare lands	**45**	0	0	8	0	3	0	0	0	56	80.4
Natural forests	0	**56**	2	0	0	0	0	0	0	58	96.6
Plantation forests	0	1	**54**	0	0	0	0	0	0	55	98.2
Croplands	2	0	0	**62**	5	0	0	0	0	69	89.9
Grasslands	0	0	0	3	**55**	0	2	1	0	61	90.2
Settlements	3	0	0	3	0	**51**	0	0	0	57	89.5
Tree patches	0	0	0	0	1	0	**62**	0	0	63	98.4
Woodlands	0	0	0	0	1	0	0	**46**	0	47	97.9
Water	0	0	0	0	0	0	0	0	**51**	51	100.0
Total	50	57	56	76	62	54	64	47	51	517	
PA(%)	90	98.2	96.4	81.6	88.7	94.4	96.9	97.9	100		

Overall accuracy = 93.2%; Kappa statistic = 0.924.

where UA = user's accuracy and PA = producer's accuracy. The columns represent actual location of samples on the ground, while rows display classified data showing location of samples in the classified images. Diagonal numbers showed in bold are the correct classifications. The off-diagonal numbers in rows and columns are misclassifications or errors.

3.1.2. States of Land Use/Land Cover (LULC)

A total of nine LULC types were extracted in the study landscape with different reference years, *i.e.*, 1973, 1986, 2000, and 2012 (Figure 3). At the beginning of the study period (1973) grasslands were the dominant LULC type, making up 42.3% of the study landscape followed by natural forests (21%), croplands (13%), woodlands (11.4%), and water (9.6%) (Table 5). Tree patches, settlements and bare lands shared small proportion of 1.9%, 0.4%, and 0.3%, respectively, of the entire landscape. In 1986, grasslands were also accounted for the largest part (37.2%) and croplands, natural forests, and water accounted 30%, 15.5% and 9.6%, respectively, of the study landscape. Woodlands, settlements, tree patches, and bare lands occupied the smallest portion of the area. In addition, plantation forests (1%) appeared during this particular reference year. In 2000, the overall situation was changed. Croplands occupied the largest portion (39.5%) of the study landscape, followed by grasslands (30.7%), natural forests (12.2%), and water (9.5%). The remaining portions were occupied by plantation forests, bare lands, settlements, and woodlands. Croplands continued to be the dominant LULC (48.5%) in 2012. Grasslands and natural forests were the second and third dominant LULC types covering 24.2% and 9.2%, respectively, of the study area. Others occupied the smallest portion of the area.

Figure 3. Land use/land cover (LULC) map of the study landscape (**a**) 1973, (**b**) 1986, (**c**) 2000 and (**d**) 2012.

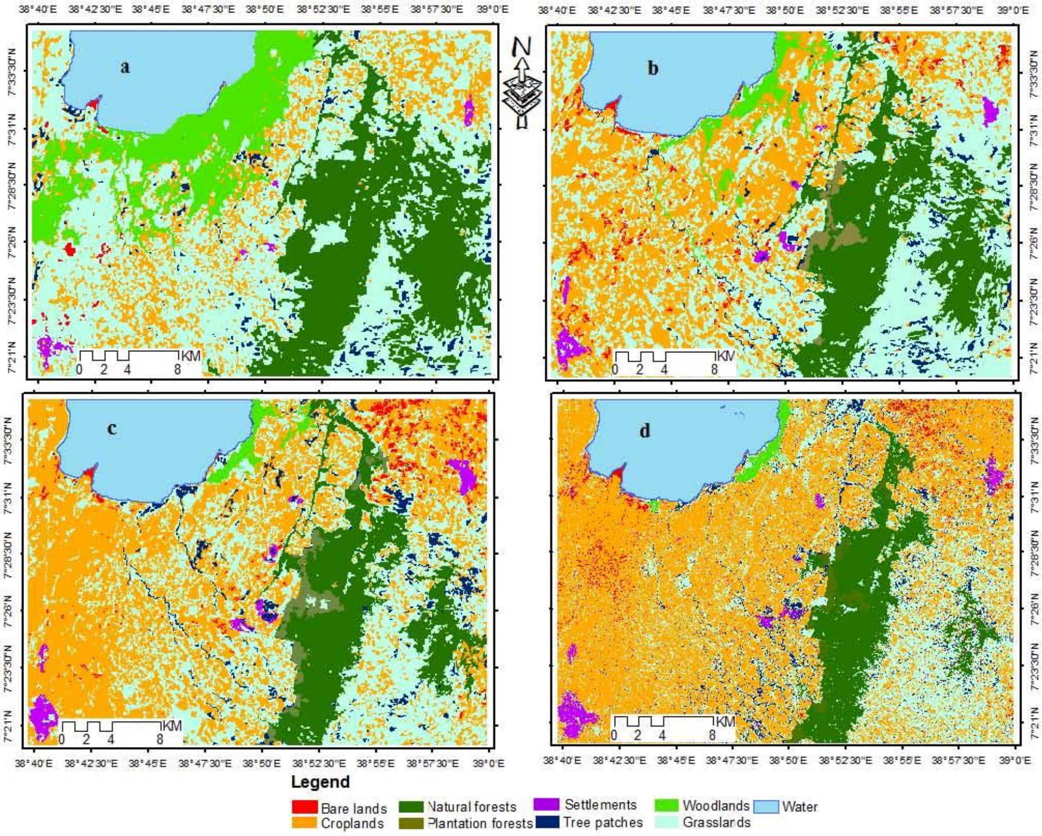


Table 5. Summaries of area of classified land use/land covers in the study area for the different reference years.

LULC Types	1973		1986		2000		2012	
	Area (ha)	(%)	Area (ha)	%	Area (ha)	(%)	Area (ha)	%
Bare lands	343	0.3	1,462	1.4	1,598	1.5	1,765	1.7
Natural forests	21,726	21	16,065	15.5	12,680	12.2	9,588	9.2
Plantation forests	-	-	1,022	1	1,707	1.6	1,284	1.2
Croplands	13,498	13	31,178	30	40,997	39.5	50,317	48.5
Grasslands	43,830	42.3	38,606	37.2	31,853	30.7	25,139	24.2
Settlements	439	0.4	867	0.8	1,139	1.1	1,586	1.5
Tree patches	2,021	1.9	2,488	2.4	2,870	2.8	3,606	3.5
Woodlands	11,842	11.4	2,150	2.1	1,026	1	656	0.6
Water	9,976	9.6	9,920	9.6	9,890	9.5	9,871	9.5

3.1.3. Land Use/Land Cover (LULC) Changes

The change results revealed a considerable reduction of woodlands, natural forests and grasslands over the first (1973–1986), second (1986–2000), and third (2000–2012) study periods. The total woodlands converted between the first study period amounts to 9,692 ha, which is about 82% of the cover that existed in 1973. The woodlands were further cleared by about 52% and 36% of the cover that existed in 1986 and 2000 during the second and third periods, respectively. Similarly, natural forests cover decreased by 26.1% during the first, 21.1% during the second, and 24.4% in the third periods. Grasslands also declined by 11.9%, 17.5% and 21.1% during the first, second, and third periods, respectively. On the contrary, croplands increased in all three periods by 131%, 31.5%, and 22.7% during the first, second and third periods. Similarly, bare lands increased by 326.2%, 9.3% and 10.5% in the three periods, respectively. Water showed a slight reduction during the whole study period (Table 6).

Table 6. Results of LULC changes from 1973 to 1986, 1986 to 2000, 2000 to 2012 and 1973 to 2012 time periods showing area changed for each classes in hectare (ha) and percentage LULC changes between periods in percentage (%).

LULC Types	Change in Land Use/Land Cover between Periods					
	1973–1986		1986–2000		2000–2012	
	Area (ha)	(%)	Area (ha)	(%)	Area (ha)	(%)
Bare lands	1,119	326.2	136	9.3	167	10.5
Natural forests	−5,661	−26.1	−3,385	−21.1	−3,092	−24.4
Plantation forests	1,022	100	685	67	−423	−24.8
Croplands	17,680	131	9,819	31.5	9,320	22.7
Grassland	−5,224	−11.9	−6,753	−17.5	−6,714	−21.1
Settlements	428	97.5	272	31.4	447	39.2
Tree patches	467	23.1	382	15.4	736	25.6
Woodlands	−9,692	−81.8	−1,124	−52.3	−370	−36.1
Water	−56	−0.6	−30	−0.3	−19	−0.2

The change matrix analysis show that as a whole, about 61,848 ha (60%) of the land within the study landscape experienced LULC changes in one or another way in the 39 years (1973–2012) study period (Table 7). The level of changes differed among the LULC types. For instance, out of 11,832.4 ha woodlands in 1973, only 650.6 ha (5.4%) remained unchanged during the study period, implying that about 95% of the woodlands were converted to other LULCs. Of the 95% converted woodland areas, 66.2% were converted for crop production. During the same period, out of 21,723.3 ha in 1973, 8,922 ha (about 41%) remained unchanged. The remaining 59% of the 1973 natural forests were converted to other LULC types. For instance, of 1,280.4 ha total cover of planation forests in 2012, about 946 ha (73.9%) was converted from the area, which used to be part of the natural forests in 1973. The original extent of grasslands (about 43,809 ha) was reduced by 74.2% due to changes to other LULCs during the period analyzed, which includes about 63.6% that was converted to croplands.

Furthermore, about 939 ha (46.5%) of the land, which used to be covered by tree patches was also converted into croplands during the same period. Major opposite trends of LULC changes were found from the change matrix analysis for croplands and settlements. Croplands replaced about 36,708 ha the land that used to be covered by other LULC types. The major conversions were from natural forests (about 3,193 ha), grasslands (about 27,875 ha), and woodlands (about 7,828 ha). As a result, croplands gained an increase of 272% during the period analyzed. There were also conversions to settlements from other LULC classes, but the one from grasslands was the highest (Table 7). About 805 ha of the land used for settlements were grasslands during the initial period of the study. Consequently, the area of settlements has also shown 260% increase from its original size in 1973.

Table 7. Summary of LULC change matrix in ha from 1973 to 2012.

From / To	Bare Lands	Natural Forests	Plantation Forests	Crop Lands	Grass Lands	Settle ments	Tree Patches	Woodlands	Water	Total (2012) [b]
Bare lands	**79**	238.6	0	310.9	922.3	1.3	16.1	187.9	5	1,761.1
Natural forests	0	**8,921.7**	0	20.0	623.8	0	14.6	0	0	9580
Plantation forest	0	945.7	**0**	6.5	326.4	0	1.9	0	0	1,280.4
Croplands	227.3	3,192.5	0	**10,072.3**	27,874.6	47.8	938.6	7,827.9	14.1	50,195.2
Grasslands	20.5	7,225.9	0	2407.7	**11,755.7**	54	889.8	2,646.4	75.4	25,075.4
Settlements	13.8	45	0	335.3	805.3	**294**	12.9	69.6	8	1,584
Tree patches	0.9	1154	0	329.4	1,491.6	42.3	**146**	425.0	9.5	3,598.7
Woodlands	0	0	0	5.1	0.1	0	0	**650.6**	0.1	655.9
Water	1.1	0	0	0.2	9.5	0	0.6	25.1	**9,834.1**	9,870.5
Total (1973) [a]	342.5	21,723.3		13,487.3	43,809.3	439.5	2,020.6	11,832.4	9,946.2	103,601.2

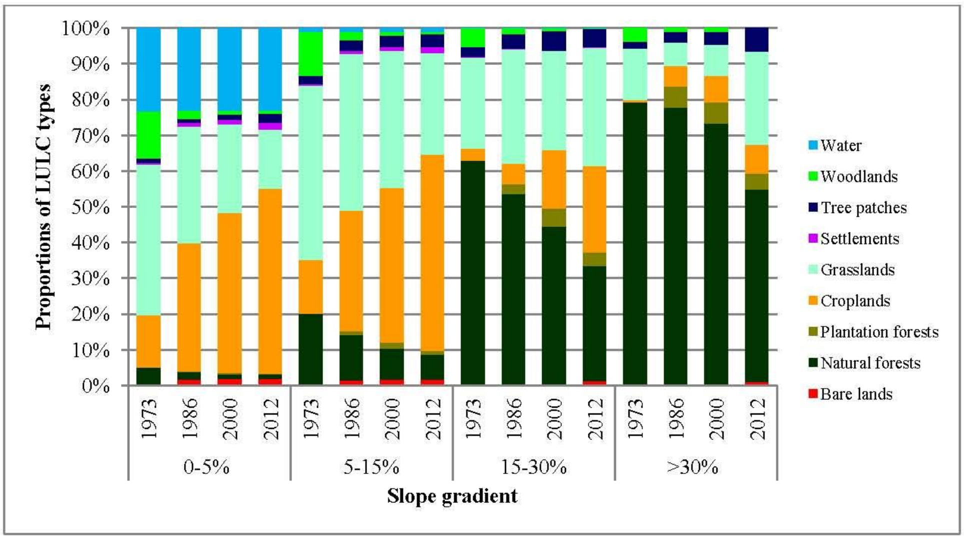
[a] Row total sums the amount of land for each LULC types of the initial study year (1973); [b] column total sums the amount of land that was converted to each LULC types of the year 2012. The values in each of the cells represent the amount of land that was converted from one land cover type to another. For instance, the value of 45 in the second column (natural forests) of the six row (settlements) means that 45 hectares of natural forests were converted to settlements from 1973 to 2012.The bold diagonal values represent the area of each class that remains unchanged while the off diagonal values represent the change area.

3.1.4. Land Use/Land Cover (LULC) Distributions and Changes along Slopes

Distribution and changes of LULC were remarkably different along the slope gradient (Figure 4). Almost all of the LULC types were found in the four slope gradients of the study landscape with different proportions. Croplands, grasslands and water were dominant LULC types in the gentle slope part. On moderate slopes, croplands, grasslands, and the natural forests were found as the dominant covers. However, the natural forest and grasslands dominated the steep and very steep slope parts of the study landscape. Other LULC types were restricted to certain slope gradients. For instance, water, settlements, and bare lands were mainly found in the gentle and moderate slope gradients. Plantation forests were limited to moderate and very steep slope parts of the study landscape.

Similar to distribution of LULC, percentages of changes were different along the slope gradient. The most significant overall trend was the shift of LULC proportions from gentle slopes (0%–5%) to very steep slopes (>30%). On steep slopes, the natural forest declined by half from about 63% in 1973 to 32% in 2012. On very steep slopes, it declined from about 79% to 54%. Woodlands also declined from about 13% to 1% on gentle slopes, 12% to 0.6% on moderate slopes, and 5% to 0.1% on steep slopes during the same study period. The area of grasslands also showed a decline from about 42% to 17% on gentle slopes, from 49% to 29% on moderate slopes and an increase from 25% to 33% on steep slopes as well as from 14% to 26% on very steep slopes. On the contrary, the area covered by croplands increased continuously from 15% to 52%, 16% to 55%, 4% to 24%, 0.6% to 8% on gentle, moderate, steep, and very steep slopes, respectively. The area covered by settlements also exhibited a triple increase on both gentle and moderate slopes of the study landscape during the same study period.

Figure 4. Proportion of LULC types in reference years along slope gradient: gentle slopes (0%–5%), moderate slopes (5%–15%), steep slopes (15%–30%), and very steep slopes (>30%). Four columns in each slope range stand for the data of 1973, 1986, 2000, and 2012 from left to right.



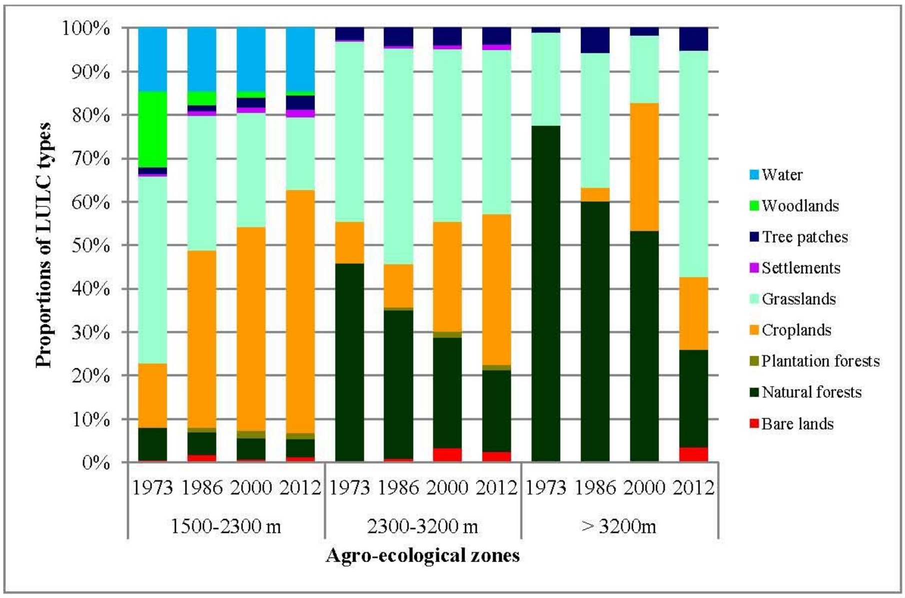
3.1.5. Land Use/Land Cover (LULC) Distributions and Changes along Agro-ecological Zones

There was a distinct relationship between land use and agro ecological zones during the study period (Figure 5). In the sub-humid highland zone (locally known as "Weyna Dega", 1,500–2,300 m), diverse LULC types were observed. All the classes of LULC types were found in this zone. Grasslands (43%), woodlands (17%), water (15%), and natural forests (8%) were the dominant LULC types at the beginning of the study period. In 2012, the dominant LULC types were croplands (56%), grasslands (17%), water (15%), and the natural forests (4%). Woodlands occupied only 1% in 2012. The numbers of LULC types were reduced in the humid highland zone ("Dega", 2,500–3,200 m). In this zone, the dominant LULC types were grasslands (42%), natural forests (46%), and croplands (10%) at the beginning of the study period. However, in 2012, the dominant LULC types were croplands (35%), grasslands (38%), and natural forests (19%). In the cold highland zone ("Wurch", >3,200 m), the natural forests (78%), grasslands (22%), and tree patches (1%) were major LULC types in 1973, and grasslands (52%), natural forests (23%), croplands (17%), and tree patches (5%) were the major LULC types in 2012.

The changes observed over the study period also differed notably along the agro-ecological zone associated with the elevation range in the study landscape. In the sub-humid highland zone, grassland showed the highest conversion from about 43% in 1973 to 17% in 2012 followed by woodlands from 17% to 1%, and natural forests from about 8% to 4%. The area covered by croplands increased from

15% to 56% during the same study period. The remaining LULC types also showed increase of cover with different magnitudes. In the humid highland zone, high conversion was observed in the natural forests followed by grasslands from about 46% to 19%, and 42% to 38%, respectively. The area of croplands was increased from 10% to 35% in this zone. In the cold highland zone, the area covered by the natural forest declined from about 78% to 23% while those of grasslands and croplands increased from about 21% to 52%, and from zero to 17%, respectively. An increase of area coverage from zero to 4% and 0.3% was also found in bare lands and settlements, respectively, in this particular zone.

Figure 5. Proportion of LULC types in reference years along Agro-ecological zone: 1,500–2,300 m = sub-humid highlands ("Weyna Dega"), 2,300–3,200 m = humid highlands ("Dega") and above 3,200 m = cold highlands ("Wurch"). Four columns in each zone stand for the data of 1973, 1986, 2000, and 2012 from left to right.



3.2. Discussion

Remote sensing data, by employing object-based image analysis, was used to provide useful information to describe LULC dynamics in the highlands of Ethiopia, particularly focusing on the Munessa-Shashemene landscape. Spatial distribution and changes of LULC over time were extracted. The overall accuracies obtained in this study were more than 91% except for Landsat MSS 1973. There were confusions among LULC types during classification of this particular image. Such confusions were associated with the lower spatial resolution and few bands available in Landsat MSS image [30,42]. However, our overall accuracies obtained from all types of images were higher than the 85% minimum threshold, set by Anderson *et al.* [59] and Thomlinson *et al.* [60] for effective LULC change analysis. Such overall classification accuracies, above the minimum threshold from the low-resolution image, were achieved by using the advantage of applying object-based classification techniques. We were able to utilize spatial relations, object features, and shapes during classification [38,40]. The overall accuracy of our object-based approach was better than other local studies with similar geographical settings, LULC types and satellite imageries. For instance, Dessie and Kleman [42] achieved 87% accuracy using Landsat TM, Wondie *et al.* [26] reported 88% accuracy using Landsat ETM+, and Shiferaw [25] achieved 86.1% using Landsat ETM+. We found above 91% over all accuracies using the same satellite imageries.

The quantitative results of change analysis of 39 years with three time periods (1973–1986, 1986–2000 and 2000–2012) and a change matrix from 1973 to 2012 revealed the extent of changes that occurred in different LULC classes throughout the four decades. In general, the area of natural forests, woodlands, grasslands and water bodies consistently decreased with varied proportions over the study periods. The reductions of these LULC types were mainly attributed to the conversion of the areas to croplands. The conversion of grasslands, natural forests, and woodlands to croplands was quite intense and common in the study landscape. Findings from elsewhere also showed that such changes are common in other areas with similar settings. For instance, after studying the LULC dynamics in the Dembecha area of northwestern Ethiopia, Zeleke and Hurni [21] stated that 99% of the forest cover in a 271 km^2 area was converted to agricultural land between 1957 and 1995. Dessie and Kleman [42] also reported conversion of more than 82% of high forests in the south-central Rift Valley of Ethiopia in about 28 years (1972–2000). A recent study on the land use and land cover dynamics in South Wollo highlands of Ethiopia [25] also revealed dramatic expansion of agricultural land and reduction of forestland, between 1972 and 2003. Knoke *et al.* [61] argued a similar increasing trend of agricultural production in order to fulfill growing food demand. Another study on land use change from 1984 to 2002 in southern Burkina Faso also revealed expansion of croplands at the expense of forest resource conversions [62]. Other local LULC studies [20,23] also indicated a similar trend. Contrary to these and our findings, increase of forest cover was observed by Bewket [22] in Chemoga watershed within the Blue Nile, which was attributed to community afforestation programs.

Our change analysis also revealed that planation forests were created at the expense of the remaining natural forest, which belongs to tropical dry Afromontane forests [45]. Cyranoski [63] reported similar conversion of natural forests to planation forests in Indonesia. Such conversion is detrimental to biodiversity conservation, unless deforestation is unavoidable and plantation forestry is a “lesser evil” [64]. An evolution of tree patches across the landscape is also linked to extensive forest fragmentation. Changes were not limited to the forest resources, but also to the grasslands and water body of the study landscape. Similar trend was also reported by Assen [65] in the East Hararghe Ethiopian Highlands.

Changes in relation to gradients of slopes show expansion of croplands from gentle and moderate slopes to steep and very steep slopes. A similar trend was also observed by Zeleke and Hurni [21] in which cultivation expanded to marginal areas as steep as >30% slope. Change analysis across agro-ecological zones also revealed that croplands largely occupied the sub-humid highland zone of the study landscape, which is the best zone for growing diverse agricultural crops [21,27]. LULC changes from natural forest to croplands in cold-highland zone were not as severe as in the sub-humid highlands since the landscape has harsh environment for diverse crop production similar to the lower agro-ecological zone [27]. Although such environment receives less pressure on the natural forest for crop production, an increase of grasslands cover was observed, which might be linked to conversion in search of additional land for grazing [66].

The continuing LULC changes in the Munessa-Shashemene landscape have different implications. As shown in the result, expansions of croplands have been met at the expense of woodlands, natural forests, and grasslands. Such kind of drastic change in land-cover can negatively alter the potential use of an area and may ultimately lead to loss of productivity [23]. Previous assessment on soil chemical and physical properties following clearing of a tropical dry Afromontane natural forest and subsequent cultivation by small holder farmers in our study landscape revealed the overall declining trends of almost all soil quality attributes in the long perspective [67]. This could, in turn, affect the local people, e.g., the 363,241 rural inhabitants [14], by reducing the means of livelihood of those who depend on land. Other implication associated with the increasing loss of the forest resource is the decline of biodiversity. According to Tesfaye *et al.* [68] the remaining dry Afromontane forests of the study landscape are dominated by indigenous trees species among which some require high conservation priority. The same authors suggested *in situ* conservation efforts for those priority species including strict protection of the remaining mother trees. The ongoing forest conversion in our study area could indirectly contribute to the loss of those high conservation priority species. A substantial loss in grassland cover in our study landscape might also lead the local people to put additional pressure on the remaining forest area for grazing their animals that have influential factor in hindering natural regeneration [68–70]. Furthermore, the ongoing pressure supports expectations of Ciais *et al.* [71] who predicted that the African forest carbon stocks would remain vulnerable. Such dynamics could change the carbon stocks and release substantial amount of carbon from the forest to the atmosphere [2].

Land cover is also one of the governing factors that determine the rate of soil loss due to erosion [22,72]. Changes in LULC could affect soil health (soil quality) or soil intactness (the ability of soils to stay in place) and increase the risk of erosion and flooding [25]. In our study landscape, Fritzsche *et al.* [44] observed the influence of erosion with minor effects under natural vegetation. Considering the LULC status of the present study area, croplands, settlements, and bare lands could be more vulnerable to soil erosion. Furthermore, the shifting of the LULC types in steep and very steep slope could worsen the situation. There is also a direct interaction between LULC changes and the hydrological process as well as the water ecology and quality of a given area [65,73–75]. Lake Alemaya, located in the lower plateau of the East Hararghe Ethiopian highland, had disappeared due to siltation as well as sediment accumulation, which is a result of deforestation [65,76]. Although not drastically altered, the water body (Lake Langano), located in the lower part of our study landscape, was consistently declining throughout the study time period. Unless appropriate measures are taken urgently, the tragic loss similar to Lake Alemaya could happen.

Topographic conditions (slope gradient and AEZs) also provide both opportunities and constraints for any use of land as found in our study landscape. A steep slope will cause more runoff and will enhance the soil erosion in the absence of vegetation cover [21]. The expansion of croplands towards marginal lands at steep slopes of our study area threatened the remaining forests, which might lead to erosion and decreasing water supplies and an overall change in the people's livelihoods. The limited number of LULC types in the very humid AEZs suggested the opportunities of not having severe conversion similar to the lower highlands. Such change analysis along slope gradient and AEZs entails to reconsider the massive program of natural resource conservation to reduce environmental degradation and poverty as well as increase agricultural productivity and food security over the last three decades undertaken by the government of Ethiopia and a consortium of donors [77], and revert the ongoing situation by devising site specific and more effective landscape management strategies.

4. Conclusions

In the landscape of Munessa-Shashemene area of the Ethiopian highlands, LULC changes have occurred in the last four decades. This was examined by object-based classification of multi-temporal remotely sensed data (Landsat and RapidEye) from four reference years followed by post classification comparisons using recent advancements of remote sensing and GIS technologies. Our approach in object-based methods increased the classification accuracy, as observed in previous studies [36]. This is new for the case of Ethiopian landscapes having diverse features and ragged topographies. The approach has the potential to be extended across other parts of the country for improved classification results. Our results from the 1973 classified image (initial period of study) revealed grasslands, woodlands and natural forests as the dominant LULC types. In 2012, croplands were the dominant LULC types followed by others in various proportions.

In this study, we were able to depict the relationship among LULC types in different time periods. A continuous increase of croplands was observed at the expense of decreasing natural forests, grasslands and woodlands. This implies that the increase of other LULC types in the area is the result of deforestation and grassland conversion. In addition, increases in tree patches along the study landscape illustrate the rapid forest fragmentation over the last four decades and the huge transformation in to monoculture agricultural systems. Such kind of agricultural landscape may lead to environmental and ecological problems. The situations were clear in our analysis of the 39-year change matrix, which revealed that about 60% of the land had experienced changes in LULC. Specifically, about 75% of the existing planation forests were established at the expense of conversion of the natural forest, which is one of the remnant tropical dry Afromontane forests in the country. Geographically, there are clear spatial patterns between the natural forest and other LULC types, whereby the natural forests dominated the steep and very steep slope parts of the study landscape. However, changes in relation to slope gradients showed expansion of croplands from gentle and moderate slopes to steep and very steep slopes. This change is perilous since it might lead to severe erosion. Agro-ecologically, croplands largely occupied the sub-humid highland zone of the study landscape. With these variations in changes along the slope gradient and agro-ecological zones, we recommend a strategic land use planning if the observed harmful trends are to be reversed.

Our findings provided up-to-date dynamics of LULC datasets. The datasets will be inputs for spatial database that can be used as a model for monitoring of future changes, informed decision making during policy formulation or land use planning process or other similar studies in the Ethiopian highlands. Furthermore, they can provide information as indicators of the direction of change in the study landscape over the given period. In this regard, our study is also in line with other conclusions about the use of remote sensing-based analysis to be a vital tool for continuous monitoring of LULC changes at varied spatial and temporal scales, which are otherwise not possible to simply attempt through ground based inventory, e.g., [25,31,34]. However, remotely sensed data help to extract information mainly on the extent of LULC changes but do not provide explanations about the reasons and drivers responsible for the changes observed. Further investigations of these remote sensing-based findings with datasets from primary or secondary sources would be helpful in finding driving forces of changes and their detailed consequences as well as looking for alternative solutions for conservation and management problems and designing future development strategies. It is important to prevent or take urgent action against further expansion of croplands to very steep slopes and harsh environment, which might have negative impacts on the remaining natural forest.

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Conflict of Interest

The authors declare no conflict of interest.

References

1. Pielke, R.A.; Marland, G.; Betts, R.A.; Chase, T.N.; Eastman, J.L.; Niles, J.O.; Niyogi, D.S.; Running, S.W. The influence of land-use change and landscape dynamics on the climate system: Relevance to climate-change policy beyond the radiative effect of greenhouse gases. *Phil. Trans. R. Soc.* **2002**, *360*, 1705–1719.
2. Intergovernmental Panel on Climate Change (IPCC). *Land Use, Land-Use Change, and Forestry: A Special Report of the Intergovernmental Panel on Climate Change*; Cambridge University Press: Cambridge, UK/New York, NY, USA, 2000.
3. Vitousek, P.M.; Mooney, H.A.; Lubchenco, J.; Melillo J.M. Human domination of Earth's ecosystems. *Science* **1997**, *277*, 494–499.
4. Mas, J.-F.; Velázquez, A.; Díaz-Gallegos, J.R.; Mayorga-Saucedo, R.; Alcántara, C.; Bocco, G.; Castro, R.; Fernández, T.; Pérez-Vega, A. Assessing land use/cover changes: A nationwide multidate spatial database for Mexico. *Int. J. Appl. Earth Obs. Geoinf.* **2004**, *5*, 249–261.
5. Prenzel, B. Remote sensing-based quantification of land-cover and land-use change for planning. *Progr. Plan.* **2004**, *61*, 281–299.
6. Giri, C.; Zhu, Z.; Reed, B. A comparative analysis of the global land cover 2000 and MODIS land cover data sets. *Remote Sens. Environ.* **2005**, *94*, 123–132.
7. Gautam, N.; Chennaiah, G. Land-use and land-cover mapping and change detection in Tripura using satellite LANDSAT data. *Int. J. Remote Sens.* **1985**, *6*, 517–528.

8. Dhinwa, P.S.; Pathan, S.K.; Sastry, S.V.C.; Rao, M.; Majumder, K.L.; Chotani, M.L.; Singh, J.P.; Sinha, R.L.P. Land use change analysis of Bharatpur district using GIS. *J. Indian Soc. Remote Sens.* **1992**, *20*, 237–250.
9. Weng, Q. Land use change analysis in the Zhujiang Delta of China using satellite remote sensing, GIS and stochastic modelling. *J. Environ. Manage.* **2002**, *64*, 273–284.
10. Rindfuss, R.R.; Walsh, S.J.; Turner, B.L.; Fox, J.; Mishra, V. Developing a science of land change: Challenges and methodological issues. *Proc. Natl. Acad. Sci. USA* **2004**, *101*, 13976–13981.
11. Nelson, G.C.; Geoghegan, J. Deforestation and land use change: Sparse data environments. *Agr. Econ.* **2002**, *27*, 201–216.
12. Poyatos, R.; Latron, J.; Llorens, P. Land use and land cover change after agricultural abandonment. *Mt. Res. Dev.* **2003**, *23*, 362–368.
13. Kidanu, S. Using Eucalyptus for Soil & Water Conservation on the Highland Vertisols of Ethiopia. Ph.D. Thesis, Wageningen University and Research Centre, Wageningen, The Netherlands, 2004.
14. Central Statistical Authority (CSA). *Summary and Statistical Report of the 2007 Population and Housing Census*; CSA: Addis Ababa, Ethiopia, 2007.
15. Eshetu, Z.; Högberg, P. Reconstruction of forest site history in Ethiopian highlands based on C^{13} natural abundance of soils. *Ambio* **2000**, *29*, 83–89.
16. Hurni, H.; Tato, K.; Zeleke, G. The implications of changes in population, land use, and land management for surface runoff in the Upper Nile Basin Area of Ethiopia. *Mt. Res. Dev.* **2005**, *25*, 147–154.
17. Lemenih, M.; Teketay, D. Potentials of plantation forests in fostering the restoration of native flora and fauna at degraded sites in Ethiopia: A review. *Ethiop. J. Biol. Soc.* **2004**, *3*, 81–111.
18. Abate, S. Land Use Dynamics, Soil Degradation, and Potential for Sustainable Use in Metu Area, Illubabor Region, Ethiopia. Ph.D. Thesis, University of Berne, Berne, Switzerland, 1994.
19. Rembold, F.; Carnicelli, S.; Nori, M.; Ferrari, G.A. Use of aerial photographs, Landsat TM imagery and multidisciplinary field survey for land-cover change analysis in the lakes region (Ethiopia). *Int. J. Appl. Earth Obs. Geoinf.* **2000**, *2*, 181–189.
20. Tekle, K.; Hedlund, L. Land cover changes between 1958 and 1986 in Kalu District, Southern Wello, Ethiopia. *Mt. Res. Dev.* **2000**, *20*, 42–51.
21. Zeleke, G.; Hurni, H. Implications of land use and land cover dynamics for mountain resource degradation in the northwestern Ethiopian Highlands. *Mt. Res. Dev.* **2001**, *21*, 184–191.
22. Bewket, W. Land cover dynamics since the 1950s in Chemoga Watershed, Blue Nile Basin, Ethiopia. *Mt. Res. Dev.* **2002**, *22*, 263–269.
23. Tegene, B. Land-cover/land-use changes in the derekolli catchment of the South Welo Zone of Amhara Region, Ethiopia. *East. Afr. Soc. Sci. Res. Rev.* **2002**, *18*, 1–20.
24. Dwivedi, R.S.; Sreenivas, K.; Ramana, K.V. Cover: Land-use/land-cover change analysis in part of Ethiopia using Landsat Thematic Mapper data. *Int. J. Remote Sens.* **2005**, *26*, 1285–1287.
25. Shiferaw, A. Evaluating the land use and land cover dynamics in Borena Wereda of South Wollo Highlands, Ethiopia. *J. Sustain. Dev. Africa* **2011**, *13*, 87–107.
26. Wondie, M.; Schneider, W.; Melesse, A.M.; Teketay, D. Spatial and temporal land cover changes in the simen mountains national park, a world heritage site in Northwestern Ethiopia. *Remote Sens.* **2011**, *3*, 752–766.

27. Hurni, H. *Agroecological Belts of Ethiopia: Explanatory Notes on Three Maps at a Scale of 1:1,000,000: Addis Ababa*; Soil Conservation Research Program Research Report; Center for Development and Environment (CDE): Addis Ababa/Berne, Switzerland, 1998.
28. Lunetta, R.S.; Knight, J.F.; Ediriwickrema, J.; Lyon, J.G.; Worthy, L.D. Land-cover change detection using multi-temporal MODIS NDVI data. *Remote Sens. Environ.* **2006**, *105*, 142–154.
29. Wu, X.; Cai, Y.; Zhou, T. Effects of land use/land cover changes on rocky desertification-a case study of a small Karst catchment in Southwestern China. *Energy Procedia* **2011**, *5*, 1–5.
30. Lillesand, T.M.; Kiefer, R.W. *Remote Sensing and Image Interpretation*, 4th ed.; John Wiley & Sons: New York, NY, USA, 2000.
31. Singh, P.; Thakur, J.K.; Kumar, S.; Singh, U.C. Assessment of Land Use/Land Cover Using Geospatial Techniques in a Semi-arid Region of Madhya Pradesh, India. In *Geospatial Techniques for Managing Environmental Resources*; Thakur, J.K., Ed.; Springer: Dordrecht, the Netherland/New York, NY, USA, 2012; pp. 152–163.
32. Richards, J.A.; Jia, X. *Remote Sensing Digital Image Analysis: An Introduction*, 3rd ed.; Springer-Verlag: Berlin, Germany/New York, NY, USA, 1999.
33. Gillanders, S.N.; Coops, N.C.; Wulder, M.A.; Gergel, S.E.; Nelson, T. Multitemporal remote sensing of landscape dynamics and pattern change: Describing natural and anthropogenic trends. *Progr. Phys. Geogr.* **2008**, *32*, 503–528.
34. Fan, F.; Weng, Q.; Wang, Y. Land use and land cover change in Guangzhou, China, from 1998 to 2003, based on Landsat TM /ETM+ Imagery. *Sensors* **2007**, *7*, 1323–1342.
35. Blaschke, T. Object based image analysis for remote sensing. *ISPRS J. Photogramm.* **2010**, *65*, 2–16.
36. Whiteside, T.G.; Boggs, G.S.; Maier, S.W. Comparing object-based and pixel-based classifications for mapping savannas. *Int. J. Appl. Earth Obs. Geoinf.* **2011**, *13*, 884–893.
37. Luo, L.; Mountrakis, G. Converting local spectral and spatial information from a priori classifiers into contextual knowledge for impervious surface classification. *ISPRS J. Photogramm.* **2011**, *66*, 579–587.
38. Baatz, M.; Schäpe, A. Multiresolution Segmentation: An Optimization Approach for High Quality Multi-scale Image. In *Angewandte Geographische Informationsverarbeitung XII: Beiträge zum AGIT-Symposium Salzburg 2000*; Blaschke, T., Griesebner, G., Strobl, J., Eds.; Herbert Wichmann Verlag: Heidelberg, Germany, 2000; pp. 12–23.
39. Manakos, I.; Schneider, T.; Ammer, U. A Comparison between the ISODATA and the eCognition Classification Methods on Basis of Field Data; In Proceedings of the XIXth ISPRS Congress, Amsterdam, The Netherlands, 16–23 July 2000.
40. Definiens. *eCognition Developer 8 Reference Book User Guide*; Definiens: Munich, Germany, 2009.
41. Dronova, I.; Gong, P.; Wang, L. Object-based analysis and change detection of major wetland cover types and their classification uncertainty during the low water period at Poyang Lake, China. *Remote Sens. Environ.* **2011**, *115*, 3220–3236.
42. Dessie, G.; Kleman, J. Pattern and magnitude of deforestation in the South Central Rift Valley Region of Ethiopia. *Mt. Res. Dev.* **2007**, *27*, 162–168.
43. Garedew, E.; Sandewall, M.; Söderberg, U.; Campbell, B.M. Land-use and land-cover dynamics in the Central Rift Valley of Ethiopia. *Environ. Manage.* **2009**, *44*, 683–694.

44. Fritzsche, F.; Zech, W.; Guggenberger, G. Soils of the Main Ethiopian Rift Valley escarpment: A transect study. *Catena* **2007**, *70*, 209–219.
45. Teketay, D.; Granström, A. Soil seed banks in dry Afromontane forests of Ethiopia. *J. Veg. Sci.* **1995**, 777–786.
46. Lu, D.; Mausel, P.; Brondízio, E.; Moran, E. Change detection techniques. *Int. J. Remote Sens.* **2004**, *25*, 2365–2401.
47. Yan, G.; Mas, J.-F.; Maathuis, B.H.P.; Xiangmin, Z.; van Dijk, P.M. Comparison of pixel-based and object-oriented image classification approaches—A case study in a coal fire area, Wuda, Inner Mongolia, China. *Int. J. Remote Sens.* **2006**, *27*, 4039–4055.
48. Tian, J.; Chen, D. Optimization in multi-scale segmentation of high-resolution satellite images for artificial feature recognition. *Int. J. Remote Sens.* **2007**, *28*, 4625–4644.
49. Flanders, D.; Hall-Beyer, M.; Pereverzoff, J. Preliminary evaluation of eCognition object-based software for cut block delineation and feature extraction. *Can. J. Remote Sens.* **2003**, *29*, 441–452.
50. Reed, B.C.; Brown, J.F.; VanderZee, D.; Loveland, T.R.; Merchant, J.W.; Ohlen, D.O. Measuring phenological variability from satellite imagery. *J. Veg. Sci.* **1994**, *5*, 703–714.
51. Teketay, D. Vegetation Types and Forest Fire Management in Ethiopia. In Proceedings of the Round Table Conference on Integrated Forest Fire Management in Ethiopia, Addis Ababa, Ethiopia,19–20 September 2000; Ministry of Agriculture (MOA)/Deutsche Gesellschaft für Technische Zusammenarbeit (GTZ): Addis Ababa, Ethiopia, 2001; pp. 1–35.
52. Foody, G.M. Status of land cover classification accuracy assessment. *Remote Sens. Environ.* **2002**, *80*, 185–201.
53. Congalton, R.G.; Green, K. *Assessing the Accuracy of Remotely Sensed Data: Principles and Practices*, 2nd ed.; CRC Press/Taylor & Francis: Boca Raton, FL, USA, 2009.
54. Singh, A. Review Article Digital change detection techniques using remotely-sensed data. *Int. J. Remote Sens.* **1989**, *10*, 989–1003.
55. Foody, G.M. Monitoring the magnitude of land-cover change around the southern limits of the Sahara. *Photogramm. Eng. Remote Sensing* **2001**, *67*, 841–847.
56. Hietel, E.; Waldhardt, R.; Otte, A. Analysing land-cover changes in relation to environmental variables in Hesse, Germany. *Landscape Ecol.* **2004**, *19*, 473–489.
57. Food and Agriculture Organization (FAO). *Guidelines for Soil Description*; FAO: Rome, Italy, 1990.
58. Ministry of Agriculture (MOA). *Agro-Ecological Zones of Ethiopia*; MoA: Addis Ababa, Ethiopia, 2000.
59. Anderson, J.R.; Hardy, E.E.; Roach, J.T.; Witmer, R.E. *A Land Use and Land Cover Classification System for Use with Remote Sensor Data*; US Geological Survey Professional Paper No. 964; USGS: Reston, WV, USA, 1976; p. 28.
60. Thomlinson, J.R.; Bolstad, P.V.; Cohen, W.B. Coordinating methodologies for scaling land cover classifications from site-specific to global. *Remote Sen. Environ.* **1999**, *70*, 16–28.
61. Knoke, T.; Román-Cuesta, R.M.; Weber, M.; Haber, W. How can climate policy benefit from comprehensive land-use approaches? *Front. Ecol. Environ.* **2012**, *10*, 438–445.
62. Paré, S.; Söderberg, U.; Sandewall, M.; Ouadba, J.M. Land use analysis from spatial and field data capture in southern Burkina Faso, West Africa. *Agr. Ecosyst. Environ.* **2008**, *127*, 277–285.

63. Cyranoski, D. Biodiversity: Logging: The new conservation. *Nature* **2007**, *446*, 608–610.
64. Brockerhoff, E.G.; Jactel, H.; Parrotta, J.A.; Quine, C.P.; Sayer, J. Plantation forests and biodiversity: Oxymoron or opportunity? *Biodivers. Conserv.* **2008**, *17*, 925–951.
65. Assen, M. Land use/cover dynamics and its implications in the dried Lake Alemaya watershed, eastern Ethiopia. *J. Sustain. Dev. Africa* **2011**, *13*, 267–284.
66. Bishaw, B. Deforestation and land degradation in the Ethiopian Highlands: A strategy for physical recovery. *Ethiop. E-J. Res. Innov. Foresight* **2009**, *1*, 5–18.
67. Lemenih, M.; Karltun, E.; Olsson, M. Assessing soil chemical and physical property responses to deforestation and subsequent cultivation in smallholders farming system in Ethiopia. *Agr. Ecosyst. Environ.* **2005**, *105*, 373–386.
68. Tesfaye, G.; Teketay, D.; Fetene, M.; Beck, E. Regeneration of seven indigenous tree species in a dry Afromontane forest, southern Ethiopia. *Flora Mor. Dist. Fun. Ecol. Plants* **2010**, *205*, 135–143.
69. Gebremedhin, B.; Pender, J.; Tesfay, G. *Collective Action for Grazing Land Management in Mixed Crop-Livestock Systems in the Highlands of Northern Ethiopia*; International Livestock Research Institute: Nairobi, Kenya, 2002.
70. Teketay, D.; Lemenih, M.; Bekele, T.; Yemshaw, Y.; Feleke, S.; Tadesse, W.; Moges, Y.; Hunde, T.; Nigussie, D. Forest Resources and Challenges of Sustainable Forest Management and Conservation in Ethiopia. In *Degraded Forests in Eastern Africa: Management and Restoration*; Bongers, F., Tennigkeit, T., Eds.; Earthscan: London, UK/Washington, DC, USA, 2010; pp. 19–64.
71. Ciais, P.; Piao, S.-L.; Cadule, P.; Friedlingstein, P.; Chédin, A. Variability and recent trends in the African terrestrial carbon balance. *Biogeosciences* **2009**, *6*, 1935–1948.
72. Reusing, M.; Schneider, T.; Ammer, U. Modelling soil loss rates in the Ethiopian Highlands by integration of high resolution MOMS-02/D2-stereo-data in a GIS. *Int. J. Remote Sens.* **2000**, *21*, 1885–1896.
73. Baron, J.S.; Hartman, M.D.; Kittel, T.G.F.; Band, L.E.; Ojima, D.S.; Lammers, R.B. Effects of land cover, water redistribution, and temperature on ecosystem processes in the South Platte Basin. *Econ. Appl.* **1998**, *8*, 1037–1051.
74. Arora, V. Modeling vegetation as a dynamic component in soil-vegetation-atmosphere transfer schemes and hydrological models. *Rev. Geophys.* **2002**, *40*, 1–25.
75. Beniston, M. Climate change in mountain regions: A review of possible impacts. *Clim. Change* **2003**, *59*, 5–31.
76. Alemayehu, T.; Furi, W.; Legesse, D. Impact of water overexploitation on highland lakes of eastern Ethiopia. *Environ. Geol.* **2007**, *52*, 147–154.
77. Kassie, M.; Zikhali, P.; Pender, J.; Köhlin, G. The economics of sustainable land management practices in the Ethiopian Highlands. *J. Agr. Econ.* **2010**, *61*, 605–627.

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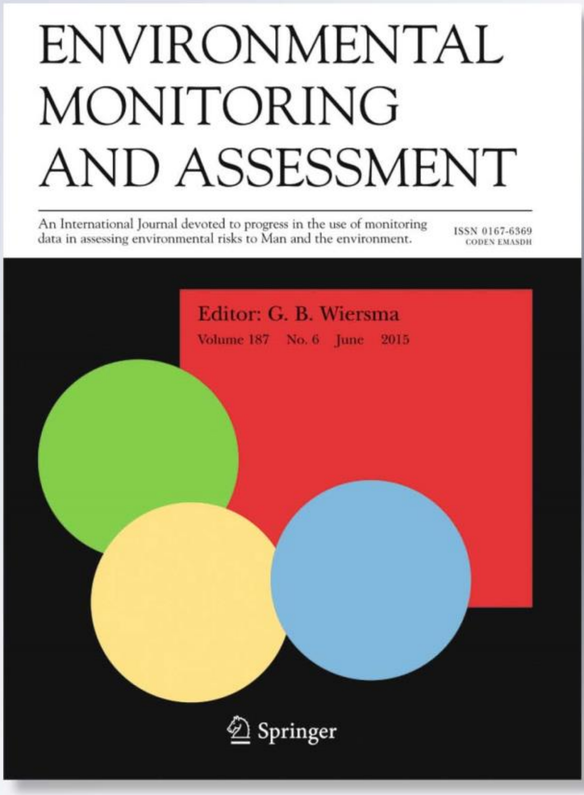

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Drivers of land use/land cover changes in Munessa-Shashemene landscape of the south-central highlands of Ethiopia

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Abstract Understanding drivers of changes in land use/land cover (LULC) is essential for modeling future dynamics or development of management strategies to ameliorate or prevent further decline of natural resources. In this study, an attempt has been made to identify the main drivers behind the LULC changes that had occurred in the past four decades in Munessa-Shashemene landscape of the south-central highlands of Ethiopia. The datasets required for the study were generated through both primary and secondary sources. Combination of techniques, including descriptive statistics, GIS-based processing, and regression analyses were employed for data analyses. Changes triggered by the interplay of more than 12 drivers were identified related to social, economic, environmental, policy/institutional, and technological factors. Specifically, population growth, expansion of cultivated lands and settlements, livestock ranching, cutting of woody species for fuelwood, and charcoal making were the top six important drivers of LULC change as viewed by the local people and confirmed by quantitative analyses. Differences in respondents' perceptions related to environmental (i.e., location specific) and socioeconomic determinants (e.g., age and literacy) about drivers were statically significant ($P = 0.001$). LULC changes were also determined by distances to major drivers (e.g., the further a pixel is from the road, the less likelihood of changes) as shown by the landscape level analyses. Further studies are suggested targeting these drivers to explore the consequences and future options and formulate intervention strategies for sustainable development in the studied landscape and elsewhere with similar geographic settings.

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Introduction

A broad range of drivers lead to land use/land cover (LULC) changes in different parts of the world (Serneels and Lambin 2001; Campbell et al. 2005; Beilin et al. 2014). Many, if not most, LULC changes are intended or unintended consequences of human decisions and the subsequent actions. The drivers of these changes may be well defined, such as population growth (Bewket 2002), pasturing (Calvas et al. 2013), urbanization (Dewan et al. 2012), or global market forces (Beilin et al. 2014), but they may also involve more complex interactions as a result of institutional or cultural influences (Campbell et al. 2005). Recently, climate-induced

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changes, such as drought, rainfall variability, and fire, have also been recognized as major drivers of LULC changes (Yang et al. 2008; Kicklighter et al. 2014; Román-Cuesta et al. 2014).

Understanding these factors that cause changes in LULC is essential for predicting future changes using models (Serneels and Lambin 2001) or development of management strategies and policies to ameliorate or prevent further decline of natural resources (Mottet et al. 2006). Although the drivers of changes are recognized as important, few studies integrate both social and ecological data from case studies to examine the links between the drivers and their effect on LULC dynamics. Yet, the trends in LULC dynamics reveal concern globally, but markedly different reasons or causes in various landscapes (Bewket 2002; Dessie and Kleman 2007; Beilin et al. 2014).

In Ethiopia, the LULC dynamics as a result of drivers shows a worrying trend (McCann 1995; Tegene 2002; Hurni et al. 2005; Kidane et al. 2012; Kindu et al. 2013; Temesgen et al. 2013). The problem is more severe in the highlands (altitude >1500 m) and covering nearly 44 % of total area of the country (Hurni et al. 2005). Different results, however, exist about drivers of LULC changes as observed from earlier studies in different parts of Ethiopia (Tekle and Hedlund 2000; Bewket 2002; Tegene 2002; Dessie and Kleman 2007; Tefera 2011). For example, Tekle and Hedlund (2000) noted expansion of settlements as causes of LULC changes at the expense of forests in South Wello of Ethiopia. Tegene (2002) reported fuelwood collection and charcoal making as major drivers for the process. Bewket (2002) claimed that population increase plays a role in LULC dynamics of Chemoga watershed within the Blue Nile area of Ethiopia. Tefera (2011) pointed out that expansion of farmlands is an important factor of LULC changes in central Ethiopia. Others considered land tenure policy as major influential factor in the observed LULC changes (McCann 1995).

While considerable efforts have been made to identify the factors, those research findings suggest that the drivers of LULC changes vary from place to place depending on location-specific factors. In addition, there is a significant debate on the level of influence by drivers of changes, and this makes generalization nearly impossible (Bewket 2002; Tegene 2002; Dessie and Kleman 2007). Thus, there is clearly a need for empirical investigation into the problem. In particular, integrated analyses of drivers on LULC dynamics is limited. As stated by Beilin et al. (2014), the drivers of changes are still contentious issues and further research is necessary. Earlier studies (Serneels and Lambin 2001; Lambin et al. 2003; Mottet et al. 2006) also underlined the need for detailed understanding of drivers that can enhance the ability to project future outcomes and intervention options. Such a comprehensive study of drivers would be useful to better understand the interrelationships between local people and the land resources, i.e., LULC types, which can help as a basis for the development of more appropriate and sustainable land use systems.

Hence, any intervention to address drivers of changes properly and development of sustainable landscape ought to begin with an empirical and locally specific understanding of the multiple drivers affecting the LULC types. This is, especially, true for the Munessa-Shashemene landscape of the Ethiopian highlands, where a tremendous LULC change has occurred (Kindu et al. 2013). In the 1970s, the dominant LULC types were forests and grasslands, but these have been overtaken mainly by croplands. All in all, about 60 % of the land showed changes in LULC. The drivers that triggered changes in this important landscape are not well understood, yet they are vital in guiding how to intervene.

Consequently, the current study emerged as the result of the quest to determine the drivers behind the LULC changes in Munessa-Shashemene landscape that can be used as a base to explore remedies or coping strategies for the ongoing problem. The challenge was to identify the drivers of LULC changes obtained through remote sensing techniques, which mainly generate the extent of LULC changes but not explanations about the underlying reasons responsible for the observed changes (Wondie et al. 2011; Kindu et al. 2013).

Therefore, in this study, we investigated, in detail, the main drivers behind the dynamics of LULC that had occurred for the past four decades in Munessa-Shashemene landscape of the south-central highlands of Ethiopia. We also examined the socioeconomic determinants of farmers' perception concerning some of the drivers at household level and explored relationship between a set of explanatory variables (drivers) and location of changes at landscape level. Ultimately, findings of this study are intended to form the basis for a better understanding of the processes of LULC changes, which researchers and policy makers could use in order to establish effective conservation, sustainable utilization, and management strategies for land resources of the studied landscape or other areas with similar settings.

Materials methods

Study area

The study area, Munessa-Shashemene landscape of the Ethiopian highlands, lies between 7° 20′ 01.23″ to 7° 35′ 13.3″N, and 38° 39′ 43.3″ to 38° 59′ 57.31″E at about 200 km south of Addis Ababa in Munessa and Arsi-Negele Districts (Kindu et al. 2013; Fig. 1). It covers an area of about 1091 km^2 and is situated between the altitudes of 1500 m above sea level at the Central Rift Valley lakes and over 3400 m at the Arsi-Bale massif. The rainfall has bimodal distribution. The short and main rainy seasons occur from March to May and June to September, respectively. Meteorological station records show that annual rainfall is about 1200 mm at Degaga town (2000 m), which is found in the study area. Mean annual temperature is 15 °C. The soils of the area are rich in clay and classified as Mazic Vertisol in the lower altitude (1500 m) and Humic Umbrisol at about 3000 m (Fritzsche et al. 2007).

It is a diverse landscape with both flat and sloped areas. About 84 % the population is rural (CSA, 2007). Crop cultivation and livestock rearing are common in all altitudinal ranges with various proportions as farmers are engaged in agriculture with a mixed farming system (Lemenih et al. 2005). Major crops grown in the area are

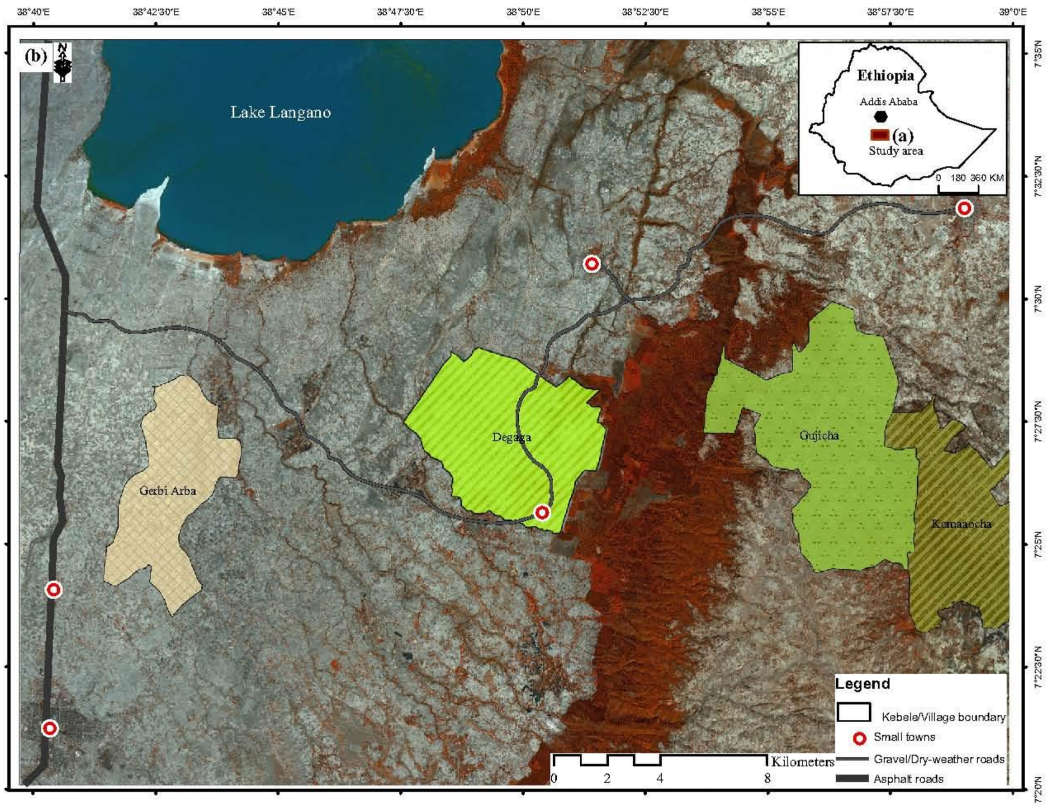

Fig. 1 Location of the study area: **a** Ethiopian context. **b** Study landscape. Background is RapidEye image of the year 2012 with RGB (*red, green, blue*): band 5, bands 4 and 3. *Polygons with different colors* are boundary of *kebeles*/villages, where data collected from local people

rain-fed barley (*Hordeum vulgare* L.), wheat (*Triticum aestivum* L.), maize (*Zea mays* L.), beans (*Phaseolus vulgaris* L.), sorghum (*Sorghum bicolor* (L.) Moench.), and teff (*Eragrostis tef* (Zucc.) Trotter.), mainly with one harvest per year. The livestock kept in the study area are, mainly, cattle and goats. They are reared in a free-grazing system. Apart from agricultural land use, the study landscape also comprises mosaics of other land use/land cover (LULC) types, mainly natural forests, plantation forests, woodlands, settlements, and water bodies. The natural forest in the area belongs to a tropical dry Afromontane forest (Teketay and Granström 1995). The plantation forests are composed of exotic species, mainly *Cupresses lusitanica* Miller, *Pinus patula* Schlechtendal & Chamisso, and *Eucalyptus* spp. Woodlands are dominated by *Acacia* spp. and found in the lower part of the study landscape. During the past four decades, conversions between these LULC types were quite intense and common in the study landscape (Kindu et al. 2013).

Data sources and methodology

The datasets required for the study were generated through both primary and secondary sources, including reports, formal household survey, group discussion with elders, field observations, and informal discussion with individual farmers and development agents (DAs) working at the study landscape. Two phases of field work were conducted. Both primary and secondary data collection and detailed household surveys, using semi-structured questionnaires, and group discussions were conducted between November 2011 and January 2012. Further field work for additional secondary data was conducted during January–February, 2013.

Sampling and household survey

The household survey was conducted in representative *kebeles* (villages) of the study landscape, which are the smallest and lowest level of administration units in Ethiopia. To generate proper data, selection of respondents was carried out in a stratified followed by two-stage sampling techniques. Accordingly, the study area was stratified in to three groups based on agro-ecology zones (AEZs), which were developed using elevation range of Aster DEM, following the categories identified by the Ministry of Agriculture (MOA 2000) and Hurni (1998). The categories include sub-humid highlands (locally known as "Weyna Dega") from 1500 to 2300 m, humid highlands ("Dega") from 2300 to 3200 m, and cold highlands ("Wurch") above 3200 m. Then, the two-stage sampling technique was applied to select the sample farm households from each of these groups. In the first stage, a purposive sampling method was employed to identify representative *kebeles* from each of the AEZs. Representative *kebeles* were selected based on information collected from a reconnaissance survey of the prevailing areas of the three agro-ecological zones, farmer interviews, agricultural expert opinion, development agents, and *kebele* administration offices. Accordingly, Gorbi Arba and Dagaga from the sub-humid highlands, Gujicha from humid highlands, and Koma Ocha from cold highlands were selected. In the second stage, sample households were randomly selected from each *kebele*. A total of 150 households were interviewed with a minimum of 30 from each *kebele*. In addition, focused group discussions were conducted in each *kebele* mainly with elders to have a clear view of drivers of changes during the study period.

The survey questionnaires covered issues regarding socioeconomic characteristics of households, drivers of LULC changes, perception of the local people, and ranking of the drivers. The selection of explanatory variables (drivers) of LULC changes incorporated in the questionnaire was based on literature and expert knowledge of the area. Other associated information, including crop production and forest and livestock situations were also explored. Similar types of issues were also covered during group discussions. Draft maps, with different reference years of the study landscape and aerial photographs, were printed on paper and served as a discussion basis to facilitate the group discussions.

Other datasets

Other datasets of the study area were obtained from multiple sources, mainly collected and synthesized from published literature, official statistics, policy documents, and on-site measurements. Population data were obtained from the Central Statistical Authority (population of 1994 and 2007) and District (Woreda) Offices (population of 2012) of the study landscape. Population estimations before and after 1994 were calculated by extrapolating the closest enumerations and the growth rate using the following formula (Bewket 2002):

$$P_2 = P_1 e^{rt} \quad (1)$$

where P_1 and P_2 = the population totals for two different time periods, t = the number of years between the two enumerations and r = the average annual rate of growth.

The LULC datasets of this study were obtained from Kindu et al. (2013). The datasets were generated by employing object-based classification of Landsat and RapidEye satellite images with overall accuracies ranging from 85.7 to 93.2 % for the different reference years. They included area statistics of different LULC covers for the year 1973, 1986, 2000, and 2012. Road network datasets were derived in ArcGIS environment from multiple sources, including Central Statistical Agency of Ethiopia, topo map, aerial photographs, and RapidEye images. GPS coordinates of market centers were collected, and information regarding each center was noted. The rainfall datasets were obtained from the Ethiopian National Meteorological Service Agency. Station records from 1981 to 2008 of Langano located in the lower part of the study landscape were used. The variables slope and agro-ecological zones (generated based on altitude) were also obtained from Kindu et al. (2013). They were derived from a 30-m digital elevation model based on Aster imagery.

Data analyses

Combinations of techniques, including descriptive statistics, GIS-based processing, and regression analyses were employed for data analyses. Descriptive statistics of simple frequency analyses were used to describe socioeconomic characteristics of households and to summarize their responses and rankings of drivers of the land use/land cover changes. Data collected through group discussions and observations were analyzed qualitatively. Association/differences in perceptions among respondents in different *kebeles* concerning drivers of LULC changes were also investigated using nonparametric test, namely Pearson's chi-square test.

Standardized rainfall anomalies and coefficient of variations were employed to evaluate inter-seasonal rainfall fluctuations. The coefficient of variation was used as statistical descriptor of rainfall variability. Standardized anomalies of rainfall of the growing season were calculated using Formula 2 and used to assess frequency of droughts as in Bewket and Conway (2007):

$$\mathrm{SRA} = \frac{(\mathrm{P}_t - \mathrm{P}_m)}{\sigma} \quad (2)$$

where SRA = standardized rainfall anomaly of growing season, P_t = growing season rainfall in year t, P_m = long-term mean rainfall of the growing season over a period of observation and σ = standard deviation of rainfall in growing season over the period of observation.

Farmers' perception of drivers for LULC changes are a function of household or socioeconomic attributes (e.g., gender, age, education, family size, occupation, or landholding size), which were generated during the survey of sample households and are summarized in Table 1. A quantitative logistic regression analysis was employed at household level to identify the main socioeconomic determinants of farmers' perception to some of the LULC driving forces for changes. The dependent variable, i.e., perception of a particular variable as a driver, is a dichotomous variable or a binary response that was generated from the questionnaire survey. On the other hand, the independent socioeconomic variables are a mixture of discrete and continuous variables. Logistic regression analysis was a suitable statistical procedure to examine the relationship between the perception (dependent) and the various socioeconomic (independent) variables, since it is an effective technique for the analyses when the dependent variable is binary, which is the case (Lesschen et al. 2005). It does not make many of the key assumptions of linear regression—particularly regarding linearity and normality. It is by nature and nonlinear and does not need a linear

Table 1 Sample households characteristics in the study landscape (N = 150)

Household attributes	Value
Interview gender (male, %)	83
Average household age (years)	47.73
Education (literate, %)	35
Household occupation (farming, %)	92
Mean household size (Nr)	7.01
Mean land holding size (ha)	2.46
Mean household income (Birr[a]/year)	32,786.79

[a] Ethiopian currency: at the time of the study, 1 USD = 17.48252 Birr

relationship between the dependent and independent variables. This logistic regression was also employed to identify the significant factors of LULC changes of the whole study landscape from 1973 to 2012 using pixels as unit of analysis, since it is an effective technique for the analysis of the process when the dependent variable is binary (Verburg et al. 2004). The dependent variable of the landscape level study was the LULC change/no change, whereas the independent variables were composed of social, economic, and environmental factors, such as population density, distance to market, distance to road, slopes, agro-ecological zones, croplands, grasslands, and bare lands. The independent variables, i.e., croplands (state of cultivated lands), grasslands (a proxy for livestock), and bare lands (indicator for degradation), are state variables meaning how much land is already covered by croplands or grasslands or bare lands during the final year of analysis (i.e., 2012).

In both cases, household and landscape level, the logistic regression function, which estimates the likelihood of the effects of the independent (explanatory) variables on the dependent (response) variable, is of the form (Lesschen et al. 2005):

$$\mathrm{Logit}(Y) = \alpha + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_3 \ldots + \beta_n X_n \quad (3)$$

where Y = the dependent variable indicating the probability that $Y = 1$, α = the intercept, $\beta_1 \ldots\ldots \beta_n$ = coefficients of the associated independent variables, and $X_1 \ldots X_n$ = the independent variables. Before the analysis, the set of independent (predictor) variables were tested for multicollinearity using a collinearity diagnostics index in linear regression analysis in SPSS (Rawlings et al. 1998). Tolerance and variance inflation factor (VIF) values are commonly utilized to screen for multicollinearity. A tolerance below 0.20 was considered a multicollinearity. It means that at least 80 % of the variance of the considered independent variable is shared with some other independent variables. A VIF of greater than 5 is generally considered evidence of multicollinearity. We also used the relative operating characteristic (ROC) to evaluate the performance of the logistic regression model (Lesschen et al. 2005). ROC is a common measure for the goodness of fit in logistic regression to know how well the independent variables correctly predict the value of the dependent variable. The ROC value of more than 0.5 is statistically better than random classifier, while ROC value of above 0.7 is acceptable for land use change modeling (Lesschen et al. 2005).

The overall approach employed in this study for landscape level analysis of drivers (Fig. 2) involved comparison of LULC changes with the potential spatially explicit explanatory variables on a cell-by-cell basis (Serneels and Lambin 2001; Verburg et al. 2004; Rutherford et al. 2007; Chen and Pontius 2010) at 60-m spatial resolution. This scale was chosen because it is related to the unit of the lowest scale for the 1973 landscape image from where the historical LULC data were generated. The datasets were prepared and changed to raster format in GIS environment at the same spatial extent and geographical coordinates. The raster datasets were, again, converted into ASCII format in GIS environment, and each ASCII data representing each variable was changed into a column format using

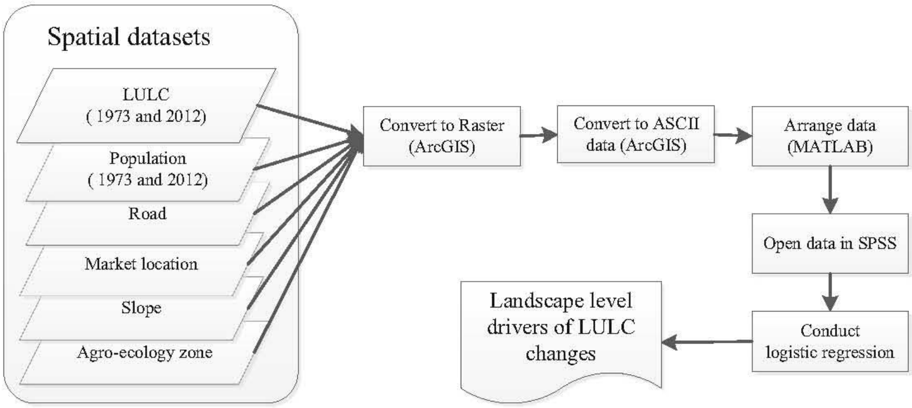

Fig. 2 Scheme used for landscape level logistic regression in the study landscape

MATLAB to make them suitable for statistical analysis. At the end, the column datasets were analyzed in SPSS using binary logistic regression to identify the relations between a set of explanatory variables (drivers) and the location of changes in LULC at landscape level.

Results

Drivers of LULC changes

A total of more than 12 factors were perceived by the respondents as being important drivers of LULC changes in the study landscape (Fig. 3). However, there were variations about each of the factors to which the local people viewed as driver for the LULC changes. In particular, all of the respondents (N = 150) perceived population growth and cultivated land expansion as the main drivers of LULC changes, but not civil war and conflict. These respondents viewed much with population and crop trends of the area. The population of the landscape has increased from 47,143 in 1973 to 233,194 in 2012 (Fig. 4), indicating an increase of 395 % during the period. A similar trend has also been observed in the area cover of croplands with an expansion of about 272 % from the area that existed in 1973 (Fig. 4). During the focused group discussions (FGDs), elders attributed these to polygamous families. There are Muslims in the study landscape that still practice having more than two wives. They also pointed out that human influxes from the neighboring areas, mainly, during the beginning the Socialist Government in Ethiopia, have intensified the pressure to the LULC change, particularly affecting forest resources negatively and increasing the area coverage of croplands. In addition, the majority of the respondents, i.e., 98.7, 97.3, and 94.7 % (N = 150), reported house construction (settlement), fuelwood collection, and charcoal making, respectively, as also important causes for the observed LULC changes in the study landscape. It was also understood from the account of the elder people during the FGDs that charcoaling is increasing over time in the study landscape.

Livestock ranching was also viewed by 94 % of the respondents as one of the important drivers for changes in the study landscape. Cattle are allowed to graze on the remaining crop stalks on the croplands after harvest and on communal grazing lands. However, with detailed discussion among the FGD members, the existing grazing land is below the carrying capacity of their livestock, which is associated to declining area of grasslands over time (Fig. 5). They pointed out that croplands and settlement expansion are responsible for the conversion. As a result, they confirmed that there are farmers who are sending their cattle to the remaining forests, including woodlands, which they also did not view as a sustainable system.

Also, land tenure by 82 %, land degradation by 76 %, improved seed variety by 73.3 %, and market access by 70.7 % of the respondents are considered to be the other factors responsible for the prevalent LULC changes.

During the FGDs, the participants described the changes in the government in 1974 (from Monarchy system to Socialist Government), which involved the

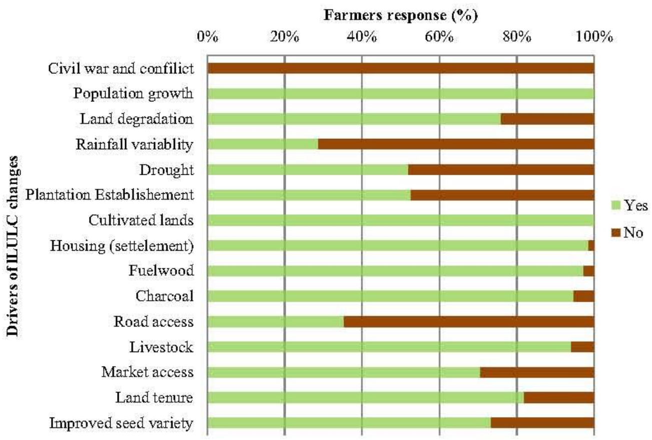

Fig. 3 Drivers of land use/land cover changes perceived by local people in Munessa-Shashemene study landscape

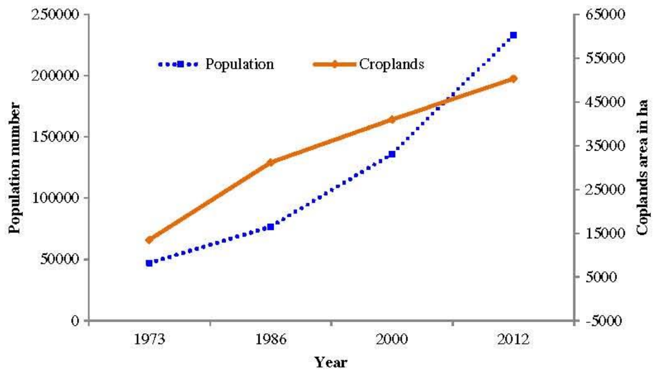

Fig. 4 Population and croplands trends from 1973 to 2012 in the Munessa-Shashemene study landscape

1975 land reform proclamation of "land to the tillers" triggered a change in land tenure from control by feudal landlords to peasant associations. This proclamation allowed peasants to have access to land for the first time, which resulted in severe LULC changes caused by the landless local people and people living in the neighboring areas. During the 1973 to 1986 study period, the 131 % expansion of croplands from that existed in 1973 (Fig. 5) implied the effect of the new land reform proclamation. The participants further explained that since the remaining forests, except plantations, are mainly state-owned, there was little effort to protect them owning poor capacity of responsible government organizations. They also added that they usually focus on meeting their immediate need rather than thinking for the future. Particularly, the increased advantage due to introduction of improved seed varieties and their associated price in combination to access to the nearby market also attracted them to expand crop production.

On the contrary, the respondents had less awareness on rainfall variability and road access as important drivers of LULC changes. In the first case, only 28.7 % of the respondents believed that rainfall variability is a factor responsible for the changes, while in the second case, only 35.3 % of the respondents considered road access as a driver of LULC changes. However, during FGDs, road access was mentioned as one of the influential drivers for changes. Similarly, the standardized growing season rainfall anomalies (SRA) in the lower part of the study landscape showed high variability in rainfall with a coefficient of variation of 30.3 %. Of the 28 years of observation, 16 years (57 %) had negative anomalies implying more dry years than wet years (Fig. 6). These show discrepancies between

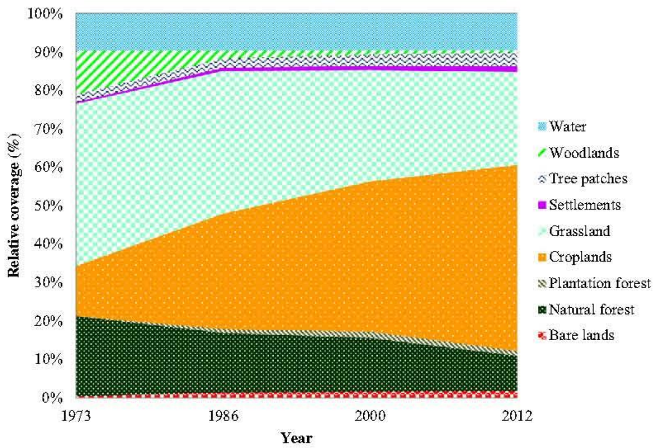

Fig. 5 Changes in land use/land cover (percentage of study area) from 1973 to 2012. *Dates* in horizontal axis represent LULC maps used to construct this time line (adopted from Kindu et al. 2013)

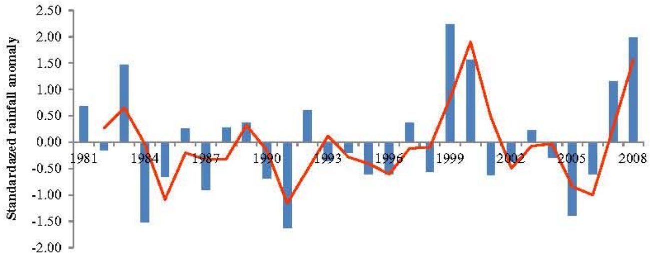

Fig. 6 Standardized anomalies of growing season rainfall in Langano station at the lower of Munessa-Shashemene study landscape for the period 1981–2008 (*red line* 2 years moving average), CV = 30.3 %

farmers' perception and what is on the ground about these two specific drivers.

Ranked drivers of changes and variations in perceptions among respondents

Farmers ranking of all the perceived drivers of LULC changes indicated that population growth was ranked first and as the most influential driver followed by expansion of cultivated lands, settlements, livestock ranching, and cutting of woody plants for fuelwood and making charcoal as well as plantation establishment, which are listed in their descending order of priority (Table 2). The highest standard deviation of ranking was observed in plantation establishment. Road access and rainfall variability were viewed as the least influential drivers of changes with rankings of 13th and 14th, respectively.

Significant differences were found in perceptions among respondents in different *kebeles* concerning land degradation, rainfall variability, drought, plantation establishment, road access, market access, and improved seed variety as drivers of LULC changes (Table 3). For instance, the majority of farmers in Degaga (96.7 %) perceived plantation establishment as a major driver of LULC changes. According to elders, the existing plantation forests were established on those lands, which were formerly covered by degraded natural forests as well as by evicting the local people who were settled and farming next to the natural forest. Contrary to that, most farmers in Gorbi Arba *kebele* did not consider plantation establishment as a key driver of change(s). Instead, the majority of these farmers perceived drought as a major driver of the observed changes. In a FGD, elders pointed out that they are dependent on selling of charcoal and fuelwood as immediate source of income during decline or failure of crop production as a result of drought years. However, most farmers in Komma Ocha did not consider drought as a major driver of changes (Table 3). Other drivers, i.e., cultivated land expansion, population

Table 2 Respondents ranked drivers of land use/land cover changes in the study landscape in order of influence (1–14), with 1 being the most influential driver

LULC drivers	Number	Min.	Max.	Mean	Std. Dev	Rank
Population growth	150	1	3	1.05	0.24	1
Cultivated lands	150	1	5	2.14	0.48	2
Housing (settlement)	148	2	5	3.30	0.61	3
Livestock	141	3	11	4.84	1.20	4
Fuel wood	146	2	10	5.23	1.20	5
Charcoal	142	3	9	5.68	1.11	6
Plantation establishment	79	1	12	5.77	3.02	7
Land tenure	123	5	13	8.20	1.82	8
Market access	106	6	12	8.42	1.06	9
Land degradation	114	1	12	8.57	1.57	10
Drought	78	2	12	9.22	2.10	11
Improved seed variety	110	5	14	10.04	1.83	12
Road access	53	3	13	10.17	2.05	13
Rainfall variability	43	7	14	10.88	1.59	14

Table 3 Association/differences between farmer's perception responses towards drivers of land use/land cover charges by *kebele*-% of respondents

LULC change drivers	Response by *kebele* Gorbi Arba (N = 30)		Dagaga (N = 30)		Gujicha (N = 50)		Kommo Ocha (N = 40)		X^2
	Yes	No	Yes	No	Yes	No	Yes	No	
Civil war and conflict	0	100	0	100	0	100	0	100	
Population growth	100	0	100	0	100	0	100	0	
Land degradation	100	0	96.7	3.3	62	38	60	40	***
Rainfall variability	56.7	43.3	40	60	20	80	10	90	***
Drought	96.7	3.3	40	60	38	62	45	55	***
Plantation establishment	30	70	96.7	3.3	64	36	22.5	77.5	***
Cultivated lands	100	0	100	0	100	0	100	0	
Housing (settlement)	100	0	100	0	98	2	97.5	2.5	
Fuel wood	100	0	100	0	98	2	92.5	7.5	
Charcoal	100	0	96.7	3.3	94	6	90	10	
Road access	63.3	36.7	46.7	53	28	72	15	85	***
Livestock	100	0	93.3	6.7	100	0	82.5	17.5	
Market access	93.3	6.7	90	10	86	14	15	85	***
Land tenure	73.3	26.7	93.3	6.7	78	22	85	15	
Improved seed variety	90	10	60	40	94	6	45	55	***

***Significant at $P < 0.001$, indicating that the location of farmers/households had a significant effect on the perception of farmers towards the drivers

growth, settlement expansion, fuelwood collection, charcoaling, livestock ranching, and insecurity of land tenure, did not show any significant variation among surveyed respondents in the different *kebeles*.

Household level logistic regression of less perceived drivers

Of the seven explanatory socioeconomic determinants (variables) included in the analysis, educational level and age of the household head (respondent) had affected positively and significantly ($P = 0.001$) the low perception of farmers on rainfall variability as drivers of LULC changes (Table 4a). Among the farmers who did not perceive rainfall variability as a driver, 77.6 and 57 % were illiterate and respondents with age lower than 50, respectively. Either literate or aged farmers were more likely to be aware of rainfall variability as a driver than the illiterate and younger farmers. The odds ratio also suggests that if a farmer is educated (literate), other factors held constant, the likelihood of awareness of the rainfall variability will be 58 times higher than an illiterate famers. Similarly, an increase in the age of a household head increases the likelihood of his/her perception of rainfall variability by a factor of 34.4. The remaining variables, namely gender and occupation of household head, family size, land holding size, and income did not contribute significantly to their perception towards rainfall as a driver of change(s).

With regard to the low perception of road access as a driver of change(s), educational level, age, and occupation of the sample household head had significant effects (Table 4b). Educational level and age of the household heads had positive effects on their perception. Those literate farmers or elders were more likely to be aware of road access as a driver than the others with odds ratio of 12.6 and 7.2 times, respectively. On the other hand, occupation had negative relationship with respondents' perception of road access as a driver of change(s). This implies that an increase in the engagement of a household head in farming activities decreases the likelihood of his/her perception of road access as a driver of change(s). The remaining variables, i.e., gender of household head, family size, land holding size, and income, did not contribute significantly to the perception of respondents concerning road access as a driver of change(s).

Table 4 Logistic regression results at household level for less perceived drivers: (a) rainfall variability and (b) road access in the study landscape

Independent variables	*B*	S.E.	Wald	*df*	Sig.	Exp(*B*)	95 % C.I. for Exp(*B*)	
							Lower	Upper
(a) Rainfall variability as a driver								
Sex (1)	−0.82	0.63	1.69	1.00	0.19	0.44	0.13	1.52
Age (1)	3.54**	1.05	11.27	1.00	0.00	34.44	4.36	71.85
Educational level (1)	4.07**	1.07	14.41	1.00	0.00	58.40	7.15	76.82
Occupation (1)	−0.91	0.84	1.17	1.00	0.28	0.40	0.77	2.1
Family size	−0.40	0.22	3.39	1.00	0.07	0.67	0.44	1.03
Land holding size	0.63	0.55	1.29	1.00	0.26	1.87	0.64	5.49
Income	0.00	0.00	0.34	1.00	0.56	1.00	0.9	1
Constant	−2.66*	1.27	4.38	1.00	0.04	0.07		
R^2	0.48							
(b) Road access as a driver								
Sex (1)	−0.70	0.62	1.28	1.00	0.26	0.50	0.15	1.67
Age (1)	1.97**	0.60	10.86	1.00	0.00	7.19	2.23	23.23
Educational level (1)	2.53**	0.61	17.03	1.00	0.00	12.56	3.78	41.77
Occupation (1)	−4.61**	1.40	10.87	1.00	0.00	0.01	0.001	0.154
Family size	−0.38	0.20	3.52	1.00	0.06	0.68	0.46	1.102
Land holding size	0.07	0.52	0.02	1.00	0.88	1.08	0.39	2.97
Income	0.00	0.00	1.13	1.00	0.29	1.00	0.99	1.00
Constant	3.22*	1.45	4.91	1.00	0.03	25.03		
R^2	0.45							

Number of observation = 150, R^2 is Negelkerke's R^2, statistically significant at $*P < 0.05$ and $**P < 0.001$. ROC (relative operation curve, i.e., correct predication) = 80 % (a). ROC = 80.7 % (b)

Logistic regression of drivers of change from 1973 to 2012 at landscape level

Results for the landscape level binary logistic regression analysis of the drivers of LULC changes in the study landscape revealed the factors, which are important drivers of changes from 1973 to 2012 in the study landscape (Table 5). The variables driving of the LULC change can be broadly categorized into three: (i) social (population density), (ii) economic (croplands, grasslands, accessibility to road, and market), and (iii) environmental (bare lands, slope, and agro-ecological zone).

The social driver, i.e., increase in population density (a proxy for growing need to more lands and fuelwood), had a significant effect ($P = 0.001$) and positive correlation with changes in LULC. The economic drivers, such as cropland expansions, grasslands (a proxy for livestock), and distance to road and market (proxies for access to farm inputs and information as well as market), had also a significant effect for the changes in LULC of the study landscape. Similarly, it is observed that the environmental drivers, i.e., slope, agro-ecological zone developed using altitudinal range, and bare lands (indicator for degradation), had also shown significant effects as drivers for the prevalent LULC changes.

Positive coefficient values of each of the drivers indicate increasing trends in the probability of LULC changes, and negative coefficient values suggest decreasing trends in the probability. From the results, a clear relation of the range of drivers can be observed with most of the signs of the coefficient values, which is in line with the hypothetical relationship. The further a pixel is from the road, the

Table 5 Landscape level logistic regression of drivers for land use/land cover changes, time period 1973–2012, unit of observation: the pixel (n = 287,988)

Variables	*B*	S.E.	Wald	*df*	Sig.	Exp(*B*)	95 % C.I. for Exp(*B*)	
							Lower	Upper
Gentle slope			3067.104	3	0			
Moderate slope	0.379	0.010	1485.318	1	0	1.460	1.433	1.489
Steep slope	0.775	0.015	2591.754	1	0	2.170	2.107	2.236
Very steep slope	0.555	0.025	506.584	1	0	1.743	1.660	1.829
Sub-humid highlands			251.721	2	0			
Humid highlands	−0.088	0.011	70.105	1	0	0.916	0.897	0.935
Cold highlands	0.532	0.044	148.262	1	0	1.702	1.562	1.854
Distance to road	−0.053	0.002	643.549	1	0	0.949	0.945	0.953
Distance to market	−0.015	0.001	318.709	1	0	0.985	0.983	0.987
Bare lands	4.050	0.069	3396.907	1	0	57.397	50.089	65.772
Croplands	2.439	0.012	40,382.828	1	0	11.465	11.195	11.741
Grasslands	1.111	0.012	8313.859	1	0	3.037	2.966	3.111
Population density	0.081	0.003	829.639	1	0	1.084	1.078	1.090
Constant	−1.007	0.020	2554.923	1	0	0.365		
R^2	0.22							

Unit of observation: pixel (n = 287,988), ROC (relative operation curve, i.e., correct predication) = 73.2 %, R^2 is Negelkerke's R^2 and statistically significant at $P < 0.001$

less likelihood of occurrence of LULC changes, and the closer a pixel is to a market, the greater the probability of occurrence of LULC changes. Also, the more the land occupied by croplands, the greater probability of occurrence of LULC changes, and the more bare land exists, the higher probability of occurrence of LULC changes. However, the results of slope coefficients were on the contrary, i.e., the greater the slope, the greater the probability of occurrence of LULC changes. The same was true for the agro-ecology variable; the higher the agro-ecological zone, the increased probability of LULC change. These imply changes progressed to marginal areas, i.e., steepy slopes and harsh agro-ecological zones, as the other areas were already exhausted (Fig. 7).

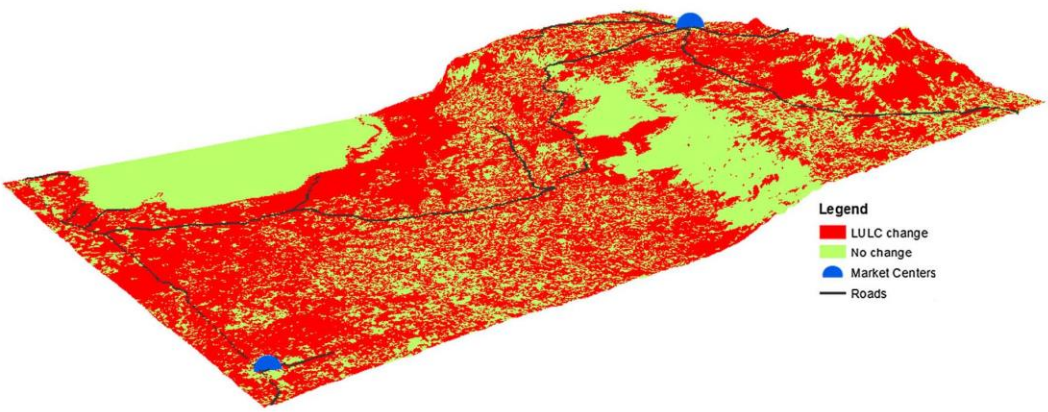

Fig. 7 3D view of land use/land cover change from 1973 to 2012, draped on Aster Digital Elevation Model (DEM)

Discussion

A range of drivers of LULC changes, which occurred for the last four decades, was identified in the study landscape. Although the level differed, local farmers perceived the majority of influential drivers of the changes. These drivers are within the broader categories of social, economic, environmental, policy/institutional, and technological forces causing the changes at national, regional, and global levels. They only varied in their levels and type of causes that trigger the changes. In the studied landscape, population growths as well as expansions of cultivated lands and settlements were the top significant drivers of change in LULC.

The population of Munessa-Shashemene landscape quadrupled in the four decades of the study periods. At the same time, to support the rising population, there was a need for extensive agricultural production with continuous crop cultivation, which was the other driver of LULC changes. This finding agrees with the report by Gashaw et al. (2014) who found that population growth and expansion of cultivated land were largely responsible for LULC changes in Northwest Ethiopia. Earlier studies in other parts of the country also reported population pressure and agricultural crops as major drivers of LULC changes (e.g., Bewket 2002; Hurni et al. 2005; Dessie and Kleman 2007; Kidane et al. 2012). Hamandawana et al. (2005) also pointed out that population negatively impacted on the environment in the form of extension of arable land into areas unsuitable for intensive agricultural utilization in Zimbabwe. Large-scale land acquisition for agricultural crop production, which is viewed as "land grabbing" also mentioned as the other contributory factor for land use changes in the global South (Woodhouse 2012; Messerli et al. 2013). In other parts of the world, e.g., Australia, Portugal, and Sweden, Beilin et al. (2014) reported global market forces as important driver for landscape dynamics. Contrary to these and our finding, Tegene (2002) found expansion of crop cultivation was not one of the primary forces for changes, which was attributed to absence of land suitable for crop production in the study site of South Welo Zone of Ethiopia during 1957–2000 study periods.

The rapid increase of the population could be partially explained by the polygamy culture practiced in the study area. Similar trends were also observed in other African countries with polygamy culture. For instance, Hayase and Liaw (1997) reported that the culture of polygamy contributed to the explosive population growth in other African countries, including, Ghana, Kenya, Senegal, and Zimbabwe. The situation seems more worrying as world population projections suggested that future population growth rates will not be uniform throughout the world. About 96 % of the additional 2.4 billion of the 7.2 billion projected to inhabit the planet in the next 40 years will be from developing countries (UN DESA 2013).

Expansion of settlements was also identified as another driver of LULC changes. The highest expansion of settlements (about 97 % of the area that existed in 1973) was found during the first study period (1973–1986). The 1985 national settlement policy for villagization program contributed for these changes (McCann 1995). Similar dynamics in LULC as a result of changes in settlement policy were also observed in other countries, including Ghana (Braimoh 2004), Kenya (Campbell et al. 2005), Tanzania (Kassahun 2011), and China (Long et al. 2008).

Fuelwood collection and charcoal making were the other important drivers of the observed LULC changes. This is directly connected with the wide use of biomass-based energy source in the country (Tegene 2002; Teketay et al. 2010; Asfaw and Demissie 2012). According to Alem et al. (2010), about 40 % of the annual charcoal supply to the Addis Ababa, the capital city of Ethiopia, is mainly from the woodlands of the Ethiopian Rift Valley, which includes the studied landscape. Having the highest calorific value (7800 Cal g^{-1}), acacia trees are preferred for charcoal making in Ethiopia (Seboka 2008), Kenya (Njenga et al. 2013), and Sudan (Khider and Elsaki 2012). This intensified the selective cuttings of the acacia-dominated woodlands that resulted in loss of the 95 % of the coverage in the studied landscape in the last four decades (Kindu et al. 2013). Additionally, in the case of decline or failure of crop production due to drought years, the local people are dependent on the sale of charcoal and fuelwood as an immediate source of income. Such coping strategies were practiced in other parts of the country (Tegene 2002; Teketay et al. 2010).

The other contributory influential driver of LULC changes was livestock ranching. Besides crop cultivation, farmers are also engaged in livestock production; hence, the farming system is, mainly, characterized as mixed crop-livestock production system. Cattle are allowed to freely graze on the remaining crop stalks on croplands after harvest and on communal grazing lands.

However, free grazing also occurs in the remaining forests due to the low carrying capacity of the available existing grazing lands. This could result in grazing-induced degradation through hindering natural regeneration of the remaining forests (Teketay et al. 2010; Tesfaye et al. 2010).

Similar to other African countries (Thiombiano and Tourino-Soto 2007), land degradation and, hence, loss of land productivity and drought that caused severe food shortage (chronic food insecurity) and the famine in 1973 and 1984 in Ethiopia were also responsible for the prevalent LULC changes (Taddese 2001). Although drought followed by food insecurity is not a new phenomenon in the country, its frequency of occurrence has increased during recent decades (Bewket and Conway (2007). In these cases, farmers living in the study landscape and the neighboring areas were forced to clear the remaining forests either for having additional croplands or maintaining their livelihoods as a source of income. This is a common survival strategy of rural populations in the events of degradation, drought, and rainfall variability across the continent (Campbell 1990). Changes in land tenure policy derived significant dynamics of LULC the country in general (McCann 1995) and to the studied landscape in particular. The land tenure policy of the country changed in 1975 due the new land reform proclamation of "land to the tillers." Consequently, landless people moved from one place to the other in search of land that resulted in additional pressure on the remaining forests (Dessie and Christiansson 2008). Presently, according to the Ethiopian constitution, land and natural resources are the property of the State and the Ethiopian people. All natural forests of the country, therefore, are not available for private ownership by law. However, the land use policy has offered the rural community the right to use the land indefinitely. Often, this serve as an incentive for farmers to convert forest lands to croplands through various means since once land is not covered with a forest, then it is possible to obtain use right over it. Though detailed research is lacking as to the extent of this phenomenon contributing to deforestation, it is clearly taking place in the forest-rich part of the country, including our study landscape (Teketay et al. 2010; Kindu et al. 2013). Campbell et al. (2005) also reported a similar incidence of changes in LULC in Kenya due to changes in land tenure policy.

Our study further revealed that among the main socioeconomic determinants, educational level (literacy), age, and occupation of household heads significantly affected their perception towards some of the drivers of LULC changes. In a case study site of eastern Ethiopia, Daba (2003) found age and literacy of households to be significant factors affecting their perception of land degradation as a problem, which is one of the drivers for the observed LULC changes. Similarly, Jadin et al. (2013) reported households that have better access to information were fully aware about drivers of forest cover dynamics in Veitnam.

Apart from household level, landscape level analyses of potential factors revealed the contribution of each driver for LULC changes in the studied area. The result showed that location of LULC changes was determined by distance to major drivers of changes. For instance, it was observed that more changes in LULC occurred in areas closer to markets and roads. The proximity of the study landscape to one of the major roads connecting Addis Ababa to the southern part of the country and Kenya and existence of the two district markets contributed to the observed LULC the changes. This finding reconfirms a conclusion by previous studies that indicated the occurrence of more LULC changes in areas where there are improved road networks (Nagendra et al. 2003; Dessie and Kleman 2007) and market access (Serneels and Lambin 2001).

Among the LULC types, the forest ecosystems, which can be used to store carbon, to protect water balance and to preserve biodiversity (Knoke and Hahn 2013) and the grasslands were largely affected by the ongoing drivers of changes. The continuing LULC dynamics as a result of drivers in the studied landscape have different implications. For instance, a substantial loss in grassland cover lead the local people to put additional pressure on the remaining forest area for grazing their animals that have influential factor in hindering natural regeneration (Teketay et al. 2010; Tesfaye et al. 2010). The ongoing forest conversion could indirectly contribute to the loss of high-conservation-priority indigenous tree species of the study landscape (Tesfaye et al. 2010). The dynamics can also negatively alter the potential use of the area and may ultimately lead to loss of productivity (Tegene 2002). In our study landscape, previous assessment on soil chemical and physical properties following clearing of a tropical dry Afromontane natural forest and subsequent cultivation by small holder farmers exposed the overall declining trends of almost all soil quality attributes in the long perspective (Lemenih et al. 2005). This could, in turn, affect the local people by reducing the means of livelihood of those who depend on land. LULC type is also one of the governing factors that determine the rate of soil loss due to erosion (Bewket 2002). Considering the LULC status of the present study area, croplands, settlements, and grasslands could be more vulnerable to soil erosion. Thus, the long-term effects of drivers of LULC dynamics in the Munessa-Shashemene landscape of the Ethiopian highlands could have negative environmental and socioeconomic consequences unless intervention strategies are introduced.

Conclusion

This study revealed that the four decades of LULC changes occurred in the Munessa-Shashemene landscape due mainly to the interplay between more than 12 drivers related to social, economic, environmental, policy/institutional, and technological factors. Specifically, population growth, expansion of cultivated lands and settlement areas (housing), livestock ranching, cutting of woody plants for fuelwood, and charcoal making are the top six important drivers of LULC changes as viewed by the local people as well as from the detailed quantitative landscape level analyses. Moreover, of all the identified drivers, significant differences among respondents living in the different *kebeles* were found with regard to land degradation, rainfall variability, drought, plantation establishment, road access, market access, and improved seed variety as drivers of LULC changes. This implies that the drivers behind the changes either were robust geographically (e.g., population growth, cultivated lands, livestock ranching) or were location specific (e.g., drought, market access, and planation development). The findings further depicted that among the main socioeconomic determinants at household level, educational level (literacy), age, and occupation of household heads significantly affected their perception towards some of the drivers of LULC changes. Consequently, either aged or literate farmers were more likely to be aware of rainfall variability as a driver, whereas an increase in the household heads engaged in farming activities decreases the likelihood of their perception of road access as drivers. At landscape level, location of LULC changes were determined by distances to major drivers of changes (e.g., the further a pixel is from the road, the less likelihood of LULC changes).

The approach employed in our study is particularly valuable in a situation like Ethiopia and other countries with similar conditions elsewhere where there are complex processes behind the dynamics of LULC types. The identified drivers can be used as inputs for modeling of future changes, informed decision making during policy formulation or land use planning processes or other similar studies in the Ethiopian highlands. Accordingly, based on the drivers identified in this study, it is important to conduct further studies to investigate the consequences and future options and formulate intervention strategies for sustainable development in the studied landscape and elsewhere with similar geographic setting. Furthermore, evidence from this study indicated that in response to the drivers, local people amplified the LULC dynamics while using the resources as a means of survival or source of income for their livelihoods. The same trends are expected to happen in the Ethiopian highlands of similar settings where subsistence farming is still present. Thus, appropriate policy and strategy centered on those major drivers is urgently required in order to avert the ongoing undesirable LULC changes of resources in this important landscape of the Ethiopian highlands.

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References

Alem, S., Duraisamy, J., Legesse, E., Seboka, Y., & Mitiku, E. (2010). Wood charcoal supply to Addis Ababa city and its effect on the environment. *Energy and Environment, 21*, 602–609.

Asfaw, A., & Demissie, Y. (2012). Sustainable household energy for Addis Ababa, Ethiopia. *Consilience: The Journal of Sustainable Development, 8*, 1–11.

Beilin, R., Lindborg, R., Stenseke, M., Pereira, H. M., Llausàs, A., Slätmo, E., Cerqueira, Y., Navarro, L., Rodrigues, P., Reichelt, N., Munro, N., & Queiroz, C. (2014). Analysing how drivers of agricultural land abandonment affect biodiversity and cultural landscapes using case studies from

Scandinavia, Iberia and Oceania. *Land Use Policy, 36*, 60–72.

Bewket, W. (2002). Land cover dynamics since the 1950s in Chemoga watershed, Blue Nile Basin, Ethiopia. *Mountain Research and Development, 22*, 263–269.

Bewket, W., & Conway, D. (2007). A note on the temporal and spatial variability of rainfall in the drought-prone Amhara region of Ethiopia. *International Journal of Climatology, 27*, 1467–1477.

Braimoh, A. K. (2004). Seasonal migration and land-use change in Ghana. *Land Degradation & Development, 15*, 37–47.

Calvas, B., Knoke, T., Castro, L. M., Hildebrandt, P., Weber, M., Stimm, B., Mosandl, R., Günter, S., & Aguirre, N. (2013). Sustainable agriculture and conservation payments are key factors in mitigating tropical forest loss. In J. Bendix (Ed.), *Ecosystem services, biodiversity and environmental change in a tropical mountain ecosystem of South Ecuador* (pp. 235–244). Berlin, New York: Springer.

Campbell, D. J. (1990). Strategies for coping with severe food deficits in rural Africa: a review of the literature. *Food and Foodways, 4*, 143–162.

Campbell, D. J., Lusch, D. P., Smucker, T. A., & Wangui, E. E. (2005). Multiple methods in the study of driving forces of land use and land cover change: a case study of SE Kajiado district, Kenya. *Human Ecology, 33*, 763–794.

Central Statistical Authority (CSA). (2007). Summary and statistical report of the 2007 population and housing census. Addis Ababa.

Chen, H., & Pontius, R. G. (2010). Diagnostic tools to evaluate a spatial land change projection along a gradient of an explanatory variable. *Landscape Ecology, 25*, 1319–1331.

Daba, S. (2003). An investigation of the physical and socioeconomic determinants of soil erosion in the Hararghe Highlands, eastern Ethiopia. *Land Degradation & Development, 14*, 69–81.

Dessie, G., & Christiansson, C. (2008). Forest decline and its causes in the south-central rift valley of Ethiopia: human impact over a one hundred year perspective. *Ambio, 37*, 263–271.

Dessie, G., & Kleman, J. (2007). Pattern and magnitude of deforestation in the south central rift valley region of Ethiopia. *Mountain Research and Development, 27*, 162–168.

Dewan, A. M., Yamaguchi, Y., & Ziaur Rahman, M. (2012). Dynamics of land use/cover changes and the analysis of landscape fragmentation in Dhaka Metropolitan, Bangladesh. *GeoJournal, 77*, 315–330.

Fritzsche, F., Zech, W., & Guggenberger, G. (2007). Soils of the main Ethiopian rift valley escarpment: a transect study. *Catena, 70*, 209–219.

Gashaw, T., Bantider, A., & Mahari, A. (2014). Population dynamics and land use/land cover changes in Dera District, Ethiopia. *Global Journal of Biology Agriculture & Health Sciences, 3*, 137–140.

Hamandawana, H., Nkambwe, M., Chanda, R., & Eckardt, F. (2005). Population driven changes in land use in Zimbabwe's district of Masvingo province: some lessons from recent history. *Applied Geography, 25*, 248–270.

Hayase, Y., & Liaw, K.-L. (1997). Factors on polygamy in Sub-Saharan Africa: findings based on the demographic and health surveys. *The Developing Economies, 35*, 293–327.

Hurni, H. (1998). Agroecological belts of Ethiopia: explanatory notes on three maps at a scale of 1:1,000,000. Addis Ababa.

Hurni, H., Tato, K., & Zeleke, G. (2005). The implications of changes in population, land use, and land management for surface runoff in the upper Nile basin area of Ethiopia. *Mountain Research and Development, 25*, 147–154.

Jadin, I., Vanacker, V., & Hoang, H. T. T. (2013). Drivers of forest cover dynamics in smallholder farming systems: the case of northwestern Vietnam. *Ambio, 42*, 344–356.

Kassahun, F. (2011). *Modelling informal settlement growth in Dar es Selaam, Tanzania*. Enschede:University of Twenete.

Khider, T. O., & Elsaki, O. T. (2012). Heat value of four hardwood species from Sudan. *Journal of Forest Products & Industries, 1*, 5–9.

Kicklighter, D. W., Cai, Y., Zhuang, Q., Parfenova, E. I., Paltsev, S., Sokolov, A. P., Melillo, J. M., Reilly, J. M., Tchebakova, N. M., & Lu, X. (2014). Potential influence of climate-induced vegetation shifts on future land use and associated land carbon fluxes in northern Eurasia. *Environmental Research Letters, 9*, 35004.

Kidane, Y., Stahlmann, R., & Beierkuhnlein, C. (2012). Vegetation dynamics, and land use and land cover change in the Bale Mountains, Ethiopia. *Environmental Monitoring and Assessment, 184*, 7473–7489.

Kindu, M., Schneider, T., Teketay, D., & Knoke, T. (2013). Land use/land cover change analysis using object-based classification approach in Munessa-Shashemene landscape of the Ethiopian highlands. *Remote Sensing, 5*, 2411–2435.

Knoke, T., & Hahn, A. (2013). Global change and the role of forests in future land-use systems. In R. Matyssek, N. Clarke, P. Cudlin, T. Mikkelsen, J.-P. Tuovinen, G. Wieser, & E. Paoletti (Eds.), *Climate change, air pollution and global challenges* (pp. 569–588). [S.l.]: Elsevier.

Lambin, E. F., Geist, H. J., & Lepers, E. (2003). Dynamics of land-use and land-cover change in tropical regions. *Annual Review of Environment and Resources, 28*, 205–241.

Lemenih, M., Karltun, E., & Olsson, M. (2005). Assessing soil chemical and physical property responses to deforestation and subsequent cultivation in smallholders farming system in Ethiopia. *Agriculture, Ecosystems & Environment, 105*, 373–386.

Lesschen, J.P., Verburg, P.H., & Staal, S.J. (2005). Statistical methods for analysing the spatial dimension of changes in land use and farming systems. International Livestock Research Institute; Wageningen University, LUCC Focus 3 Office, Nairobi, Wageningen, 80 pp.

Long, H., Wu, X., Wang, W., & Dong, G. (2008). Analysis of urban-rural land-use change during 1995–2006 and its policy dimensional driving forces in Chongqing, China. *Sensors, 8*, 681–699.

McCann, J. (1995). *People of the plow. An agricultural history of Ethiopia, 1800–1990*. Madison:University of Wisconsin Press 298 pp.

Messerli, P., Heinimann, A., Giger, M., Breu, T., & Schönweger, O. (2013). From 'land grabbing' to sustainable investments in land: potential contributions by land change science. *Current Opinion in Environmental Sustainability, 5*, 528–534.

Ministry of Agriculture (MOA) (2000). *Agro-ecological zones of Ethiopia*. Addis Ababa:Natural Resource Management and Regulatory Department.

Mottet, A., Ladet, S., Coqué, N., & Gibon, A. (2006). Agricultural land-use change and its drivers in mountain landscapes: a case study in the Pyrenees. *Agriculture, Ecosystems & Environment, 114*, 296–310.

Nagendra, H., Southworth, J., & Tucker, C. (2003). Accessibility as a determinant of landscape transformation in western Honduras: linking pattern and process. *Landscape Ecology, 18*, 141–158.

Njenga, M., Karanja, N., Munster, C., Iiyama, M., Neufeldt, H., Kithinji, J., & Jamnadass, R. (2013). Charcoal production and strategies to enhance its sustainability in Kenya. *Development in Practice, 23*, 359–371.

Rawlings, J. O., Pantula, S. G., & Dickey, D. A. (1998). *Applied regression analysis. A research tool* (2nd ed.). New York: Springer.

Román-Cuesta, R. M., Carmona-Moreno, C., Lizcano, G., New, M., Silman, M., Knoke, T., Malhi, Y., Oliveras, I., Asbjornsen, H., & Vuille, M. (2014). Synchronous fire activity in the tropical high Andes: an indication of regional climate forcing. *Global Change Biology, 20*, 1929–1942.

Rutherford, G. N., Guisan, A., & Zimmermann, N. E. (2007). Evaluating sampling strategies and logistic regression methods for modelling complex land cover changes. *Journal of Applied Ecology, 44*, 414–424.

Seboka, Y. (2008). Charcoal supply chain study in Ethiopia. In R. Kwaschik (Ed.), *Proceedings of the "Conference on charcoal and communities in Africa"* (pp. 53–61).

Serneels, S., & Lambin, E. F. (2001). Proximate causes of land-use change in Narok district, Kenya: a spatial statistical model. *Agriculture, Ecosystems & Environment, 85*, 65–81.

Taddese, G. (2001). Land degradation: a challenge to Ethiopia. *Environmental Management, 27*, 815–824.

Tefera, M. M. (2011). Land-use/land-cover dynamics in Nonno District, Central Ethiopia. *Journal of Sustainable Development in Africa, 13*, 123–141.

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

Teketay, D., & Granström, A. (1995). Soil seed banks in dry Afromontane forests of Ethiopia. *Journal of Vegetation Science, 777–786.*

Teketay, D., Lemenih, M., Bekele, T., Yemshaw, Y., Feleke, S., Tadesse, W., Moges, Y., Hunde, T., & Nigussie, D. (2010). Forest resources and challenges of sustainable forest management and conservation in Ethiopia. In F. Bongers, & T. Tennigkeit (Eds.), *Degraded forests in Eastern Africa. Management and restoration* (pp. 19–64). London, Washington, DC: Earthscan.

Tekle, K., & Hedlund, L. (2000). Land cover changes between 1958 and 1986 in Kalu District, Southern Wello, Ethiopia. *Mountain Research and Development, 20*, 42–51.

Temesgen, H., Nyssen, J., Zenebe, A., Haregeweyn, N., Kindu, M., Lemenih, M., & Haile, M. (2013). Ecological succession and land use changes in a lake retreat area (Main Ethiopian Rift Valley). *Journal of Arid Environments, 91*, 53–60.

Tesfaye, G., Teketay, D., Fetene, M., & Beck, E. (2010). Regeneration of seven indigenous tree species in a dry Afromontane forest, southern Ethiopia. *Flora - Morphology, Distribution, Functional Ecology of Plants, 205*, 135–143.

Thiombiano, L., & Tourino-Soto, I. (2007). Status and trends in land degradation in Africa. In M. V. K. Sivakumar (Ed.), *Climate and land degradation* (pp. 39–53). Berlin [u.a.]: Springer.

United Nations, Department of Economic and Social Affairs (UN DESA) Population Division (2013). *World Population Prospects. The 2012 Revision, Highlights and Advance Tables*. New York:United Nations.

Verburg, P. H., Schot, P. P., Dijst, M. J., & Veldkamp, A. (2004). Land use change modelling: current practice and research priorities. *GeoJournal, 61*, 309–324.

Wondie, M., Schneider, W., Melesse, A. M., & Teketay, D. (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.

Woodhouse, P. (2012). New investment, old challenges. Land deals and the water constraint in African Agriculture. *Journal of Peasant Studies, 39*, 777–794.

Yang, Y., Feng, Z., Huang, H. Q., & Lin, Y. (2008). Climate-induced changes in crop water balance during 1960–2001 in Northwest China. *Agriculture, Ecosystems & Environment, 127*, 107–118.

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Changes of ecosystem service values in response to land use/land cover dynamics in Munessa–Shashemene landscape of the Ethiopian highlands

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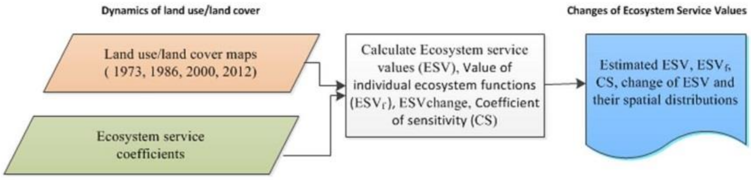
HIGHLIGHTS

- Estimating ecosystem service value changes provided insights into ecosystem status.
- We estimate at least US$ 19.3 million loss of service values in four decades.
- Huge conversion of forests is still affecting provision of ecosystem services.
- Ongoing crop production system affected ecosystem health.
- Decline of ecosystem service values reflected effects of ecological degradation.

GRAPHICAL ABSTRACT



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ABSTRACT

Land use/land cover (LULC) dynamics alter ecosystem services values (ESVs), yet quantitative evaluations of changes in ESVs are seldom attempted. Using Munessa–Shashemene landscape of the Ethiopian highlands as an example, we showed estimate of changes in ESVs in response to LULC dynamics over the past four decades (1973–2012). Estimation and change analyses of ESVs were conducted, mainly, by employing GIS using LULC datasets of the year 1973, 1986, 2000 and 2012 with their corresponding global value coefficients developed earlier and our own modified conservative value coefficients for the studied landscape. The results between periods revealed a decrease of total ESVs from US$ 130.5 million in 1973, to US$ 118.5, 114.8 and 111.1 million in 1986, 2000 and 2012, respectively. While using global value coefficients, the total ESVs declined from US$ 164.6 million in 1973, to US$ 135.8, 127.2 and 118.7 million in 1986, 2000 and 2012, respectively. The results from the analyses of changes in the four decades revealed a total loss of ESVs ranging from US$ 19.3 million when using our own modified value coefficients to US$ 45.9 million when employing global value coefficients. Changes have also occurred in values of individual ecosystem service functions, such as erosion control, nutrient cycling, climate regulation and water treatment, which were among the highest contributors of the total ESVs. However, the value of food production service function consistently increased during the study periods although not drastically. All in all, it must be considered a minimum estimate of ESV changes due to uncertainties in the value coefficients used in this study. We conclude that the decline of ESVs reflected the effects of ecological degradation in the studied landscape and suggest further studies to explore future options and formulate intervention strategies.

1. Introduction

Ecosystem services are the direct and indirect contributions of ecosystems to human wellbeing and survival (Millennium Ecosystem Assessment (MEA), 2005; Schägner et al., 2013; Costanza et al., 2014).

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Quantifying and analyzing changes of ecosystem service values (ESVs) is an important tool to raise awareness (Liu et al., 2010), contribute to developing knowledge on management of natural capital (Costanza et al., 1997; Frélichová et al., 2014), improve decision making for allocation of scarce resources among competing demands (Guo et al., 2001; Barral and Oscar, 2012), formulate polices (Schägner et al., 2013) and provide a stimulus to conserve the ecosystems that offer the most valuable services (Konarska et al., 2002; Bateman et al., 2013). As a result, interest in ecosystem service values has evolved rapidly in both the scientific communities and policy (Turner et al., 2003; Troy and Wilson, 2006; Butler et al., 2013; Costanza et al., 2014).

The provision of ecosystem services is directly correlated to the situation of ecosystems, e.g. land use/land cover (LULC) types, in a given area (de Groot et al., 2002; Styers et al., 2010). Dynamics of LULC can cause changes in the values of ecosystem services (Kreuter et al., 2001; Hu et al., 2008; Polasky et al., 2011). It may increase the provision of some services while decreasing others that affect the ability of biological systems to support human needs, indicating ecological degradation (Polasky et al., 2011), or may cause vice versa. As changes in ESVs differ depending on the direction, and/or magnitude of the LULC dynamics, most of the available studies were location specific. For instance, Leh et al. (2013) revealed a general decline of ESVs while Wang et al. (2014) found the opposite, i.e. an increasing trend. Consequently, making generalization of the results to other areas might lead to erroneous conclusions.

The Ethiopian highlands (altitude > 1500 m) that cover nearly 44% of total landmass of the country (Hurni et al., 2005); have been occupied by the majority of the people owing to availability of more favorable environmental conditions for agriculture as well as human and animal health than the lowlands (Eshetu and Högberg, 2000; Lemenih and Teketay, 2005). The pressure on these highland ecosystems has, therefore, been intense, leading to a worrying trend in terms of LULC changes (Hurni et al., 2005; Dessie and Kleman, 2007; Kindu et al., 2013). Even at present, the conversion of land for the production of crops is continuing at an increasing trend (Kidane et al., 2012; Biazin and Sterk, 2013; Kindu et al., 2013, 2015; Temesgen et al., 2013). However, beyond analyzing LULC dynamics, a thorough understanding of the consequences of these changes on ecosystem service values through quantitative knowledge is limited. This includes deriving coefficients for ecosystem service values, with a higher local validity for Ethiopia, but also more general issues, such as addressing spatial variation in ecosystem service values as well as possible variation of ecosystems service values over time.

Quantifying the ESVs and their changes has received wide attention after the publication of Costanza et al. (1997), who proposed a list of ecosystem service value coefficients of biomes (LULC types) and estimates of global ESVs. Although the proposed value coefficients have been criticized because of uncertainties (Limburg et al., 2002; Hein et al., 2006; Maes et al., 2012), a number of researchers working in regions where data are scarce have used them and paved the path to the science of ecosystem service valuation (Kreuter et al., 2001; Zhao et al., 2004; Wang et al., 2006; Li et al., 2007; Hu et al., 2008; Kubiszewski et al., 2013). Remote sensing and GIS technologies were commonly applied in most of the studies during the estimation process. The former offers opportunities of generating LULC types for a given area that can be utilized as proxies of measurements while the latter is used for estimating and mapping their distributions (Konarska et al., 2002; Zhao et al., 2004; Wang et al., 2006). As primary datasets are expensive, or sometimes scarce in some regions, secondary data consisting of spatial units, such as LULC classes, are also, more often, used as proxies for estimation (Kreuter et al., 2001; Maes et al., 2012). In addition, for the corresponding value coefficients, The Economics of Ecosystems and Biodiversity (TEEB) valuation database was also developed mainly based on literature of case studies in different parts of the globe (van der Ploeg and de Groot, 2010).

While considerable efforts have been made to estimate ESVs and suggest options for areas with scarce data, such studies lack for many countries in Africa, including Ethiopia, that have dramatic LULC dynamics (Leh et al., 2013). This is also true for the Munessa–Shashemene landscape of the Ethiopian highlands, where a rapid LULC change is occurring (Kindu et al., 2013). In the 1970s, the dominant LULC types were forests (i.e. natural forests and woodlands) and grasslands, but these have mainly been replaced by croplands. About 60% of the land showed changes in LULC types. The effects of such dynamics on ecosystem service values in this important landscape are not well understood although they are vital in raising awareness of the consequences and improved decision making. Therefore, the aims of this study are: 1) To test the variation of changes in ESV through more locally valid coefficients, when compared to the often used global coefficients published by Costanza et al. (1997). This will be achieved by an estimation of changes in ecosystem service values in response to LULC dynamics that occurred over the past four decades (1973–2012) in the Munessa–Shashemene landscape of the Ethiopian Highlands. 2) To explore the contribution and changes of individual service functions and effects of dynamics in each LULC type on changes of the corresponding ecosystem service values. 3) To carry out sensitivity studies to explore the robustness of the results. This includes a 50% adjustment of value coefficients and analysis of the impact of a variation of the value coefficients over time, which might be imposed by changes in the consumer and the producer price indices. 4) To discuss the possible impact of spatial variation in ESV coefficients.

2. Materials and methods

2.1. Study area

Munessa–Shashemene landscape of the Ethiopian highlands was selected as the study area. The landscape is located within Munessa and Arsi-Negele Districts (between 7°20′01.23″ to 7°35′13.3″N and 38°39′ 43.3″ to 38°59′57.31″E) about 200 km south of Addis Ababa (Fig. 1). It is characterized by diverse topographic conditions. The elevation ranges from 1500 m at the Central Rift Valley lakes to over 3400 m at the Arsi-Bale massif. A mountainous and undulating topography with steep slopes characterizes the eastern part of the landscape, and gentle to moderate slopes characterizes the western part. The rainfall has a bimodal distribution. The short and main rainy seasons occur from March–May and July–September, respectively. Meteorological station records show that the annual rainfall is about 1200 mm at Degaga town (2000 m), which is found in the study area. Mean annual temperature is 15 °C. The soils of the area are rich in clay and classified as Mazic Vertisol in the lower altitude (1500 m) and Humic Umbrisol at about 3000 m (Fritzsche et al., 2007).

The total area of the landscape is about 1091 km^2, and it is inhabited by about 84% rural population (Central Statistical Authority (CSA), 2007). Crop cultivation and livestock rearing is common in all altitudinal ranges with various proportions since farmers are engaged in agriculture with a mixed farming system (Lemenih et al., 2005). Apart from croplands, the study landscape also comprises mosaics of other LULC types, mainly natural forests, plantation forests, woodlands, settlements and water bodies. The natural forest in the area belongs to the tropical dry Afromontane forest (Teketay and Granström, 1995). The plantation forests are composed of exotic species, mainly *Cupresses lusitanica* Miller, *Pinus patula* Schlechtendal & Chamisso and *Eucalyptus* spp. Woodlands are dominated by *Acacia* spp. and found in the lower altitude of the study landscape. During the past four decades, conversions between these LULC types were quit intense and common in the study landscape (Kindu et al., 2013).

2.2. Data used

Datasets required for this study were obtained from various sources (Tables 1 and 2). The LULC datasets were obtained from Kindu et al. (2013). The datasets were generated by employing object-based

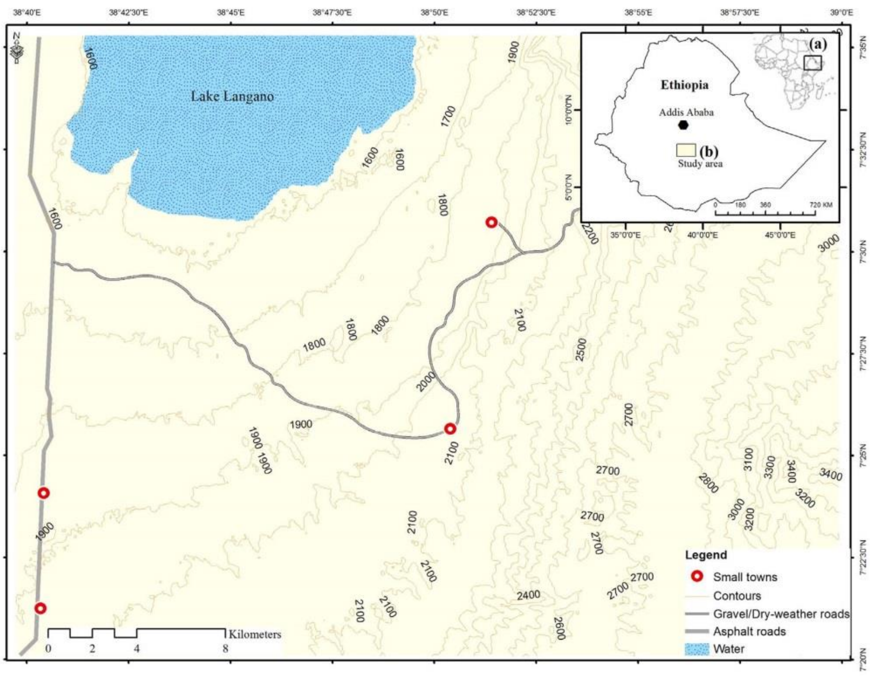

Fig. 1. Location of the study area. (a) African context, (b) landscape level study location in reference to Ethiopia. Topographic counters with 100 m were developed from Aster Digital Elevation Model.

classification of Landsat and RapidEye satellite images with different reference years. They included area statistics of nine different LULC types for the year 1973, 1986, 2000 and 2012. Two types of ecosystem service value coefficients of the target LULC types were used for this study. The first were global coefficients adopted only from Costanza et al. (1997) ecosystem service value coefficients using representative biome as a proxy for each LULC type. They proposed and employed in their ecosystem service valuation model 17 types of ecosystem service value coefficients for 16 biomes (LULC types). In our study, each of the nine LULC types of the different reference years were compared with those representative biomes (LULC types) in order to obtain their corresponding ecosystem service value coefficients identified in Costanza et al. (1997). A summary is given in Table 1. The second types of coefficients were more conservative and modified from those employed by Costanza et al. (1997) using expert knowledge of the study landscape conditions and other studies, mainly from the Economics of Ecosystems and Biodiversity (TEEB) valuation database (van der Ploeg and de Groot, 2010) and (Knoke et al., 2011). Modification was done through a benefit transfer method, which refers to the process of using existing values and other information from the original study site to estimate ESVs of other similar location in the absence of site-specific valuation information (Kreuter et al., 2001; Kubiszewski et al., 2013). In order to ensure applicability of the transferred data from TEEB valuation database to the studied landscape conditions, we only considered values from tropical areas of LULC types similar to our geographical setting. The TEEB valuation database contained more than 1300 original values compiled mainly based on local studies across the world (van der Ploeg and de Groot, 2010). We followed the same strategy as for using values from Knoke et al. (2011). Table 2 gives details of modified annual value coefficients for ecosystem services of each LULC types. Both approaches of directly using the available global value coefficients or their modification have been applied by a number of researchers for similar studies

Table 1
Summary of land use/land cover (LULC) types with their area in ha for 1973, 1986, 2000 and 2012 (adopted from Kindu et al. (2013)), and biome equivalents with the corresponding value coefficients (1994 US$ ha^{-1} $year^{-1}$): (a) global value coefficients adopted from Costanza et al. (1997) and (b) own modified conservative value coefficients mainly based on the Economic of Ecosystem and Biodiversity valuation database (van der Ploeg and de Groot, 2010) and Knoke et al. (2011).

LULC types	Area (ha)					US$ (ha^{-1} $year^{-1}$)	
	1973	1986	2000	2012	Equivalent biome	a	b
Bare lands	343	1462	1598	1765	Desert	0	0
Natural forests	21,726	16,065	12,680	9588	Tropical forest	2008	986.69
Plantation forests	0	1022	1707	1284	Tropical forest	2008	986.69
Croplands	13,498	31,178	40,997	50,317	Cropland	92	225.56
Grasslands	43,830	38,606	31,853	25,139	Grass/rangelands	244	293.25
Settlements	439	867	1139	1586	Urban	0	0
Tree patches	2021	2488	2870	3606	Grass/rangelands	244	293.25
Woodlands	11,842	2150	1026	656	Tropical forest	2008	986.69
Water	9976	9920	9890	9871	Lakes/rivers	8498	8103.5

Table 2
Details of own modified conservative annual value coefficients for ecosystem service functions of each LULC type mainly based on the Economic of Ecosystem and Biodiversity valuation database (van der Ploeg and de Groot, 2010) and Knoke et al. (2011).

Ecosystem services	Each LULC types ecosystem service values (US$/ha/yr)						
	Natural forests	Plantation forests	Croplands	Grasslands	Tree patches	Woodlands	Water
Provisioning services							
Water supply	8	8				8	2117
Food production	32	32	187.56	117.45	117.45	32	41
Raw material	51.24	51.24				51.24	
Genetic resources	41	41				41	
Regulating services							
Water regulation	6	6		3	3	6	5445
Water treatment	136	136		87	87	136	431.5
Erosion control	245	245		29	29	245	
Climate regulation	223	223				223	
Biological control			24	23	23		
Gas regulation	13.68	13.68		7	7	13.68	
Disturbance regulation	5	5				5	
Supporting services							
Nutrient cycling	184.4	184.4				184.4	
Pollination	7.27	7.27	14	25	25	7.27	
Soil formation	10	10		1	1	10	
Habitat/refugia	17.3	17.3				17.3	
Cultural services							
Recreation	4.8	4.8		0.8	0.8	4.8	69
Cultural	2	2				2	
Total	986.69	986.69	225.56	293.25	293.25	986.69	8103.5

in data scarce areas (e.g. Kreuter et al., 2001; Zhao et al., 2004; Wang et al., 2006; Li et al., 2007; Hu et al., 2008; Kubiszewski et al., 2013). All the value coefficients were converted into 1994 US$ per hectare per year to facilitate the estimation process of ESV changes. The value coefficients were also adjusted using consumer price index and producer price index (http://data.imf.org/; http://faostat.fao.org/) to check the effect of time development in the coefficients on the overall estimation of ESV changes.

National average per capita gross domestic product (GDP) of 1973 and 2012 with the corresponding population of the studied landscape was used to estimate GDP of the landscape. Other datasets, such as roads, small towns and topographic counters were utilized for study site map preparation. The small towns and road network datasets were derived in GIS environment from multiple sources, including topographical maps, aerial photographs and RapidEye images. The topographic counters developed from a 30 m Digital Elevation Model (DEM) based on Aster imagery.

2.3. Data analyses

The overall approach employed in this study (Fig. 2) involved estimating ESVs for 1973, 1986, 2000 and 2012 reference years, mapping their spatial distribution and computing the changes between study periods. The LULC datasets of each reference year to be used as a proxy for

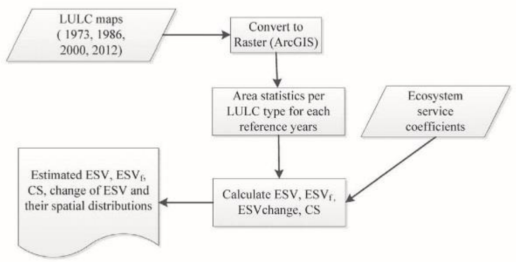

Fig. 2. Methodological approaches in the study landscape. ESV_f: estimated value of individual ecosystem functions (ESV_f); ESV: estimated total ecosystem service values for each reference year; CS: coefficient of sensitivity (CS).

the measurement of ESVs were prepared and the corresponding area in hectare was summarized in the GIS environment. In the ecosystem service estimation process, the value coefficients were assigned to each LULC type according to the value used by Costanza et al. (1997) and own modified coefficients (Table 1). Then, the area of each LULC type in hectare was multiplied by its corresponding value coefficients to calculate the total ecosystem service value for a particular LULC type. The values for the LULC types in each reference year were summed to estimate total ESV of the landscape for each reference year.

$$ESV = \sum(A_k \times VC_k \qquad (1)$$

where ESV = total estimated ecosystem service value, A_k = the area (ha) and VC_k = the value coefficient (US$ ha^{-1} $year^{-1}$) for LULC type 'k'.

The changes of ESV were obtained by calculating the difference between the estimated values in each reference year. ESV change statistics were computed by comparing values of one dataset with the corresponding value of the second dataset in each period. This resulted in a summary table of the overall changes in ESV. The values were presented as US$ and percentages. The percentage ESV changes were calculated using the following equation:

$$\text{Percentage ESV change} = \left(\frac{ESV_{\text{final year}} - ESV_{\text{initial year}}}{ESV_{\text{initial year}}}\right) \times 100 \qquad (2)$$

where ESV = total estimated ecosystem service value. Positive values suggest an increase whereas negative values imply a decrease in amount. Moreover, we also estimated values of services provided by individual ecosystem functions within the study landscape using the following equation:

$$ESV_f = \sum(A_k \times VC_{f\,k}) \qquad (3)$$

where ESV_f = calculated ecosystem service value of function 'f', A_k = the area (ha) and VC_{fk} = value coefficient of function 'f' (US$ ha^{-1} $year^{-1}$) for LULC type 'k'. The contributions of individual ecosystem functions to the overall value of ecosystem services per year were ranked based on an estimated value of ecosystem functions for each reference year and summarized in table.

Considering uncertainties exist in the value coefficients and since the biomes used as proxies for LULC types are not perfect matches in every case, sensitivity analyses were conducted to determine the percentage change in ESVs for a given percentage change in the value coefficient (Li et al., 2010). Accordingly, the ecosystem modified value coefficients for natural forest, plantation forest, croplands, grasslands, tree patches, woodlands, and water bodies were each adjusted by 50% and the corresponding coefficient of sensitivity (CS) was calculated using Eq. (4) as in Kreuter et al. (2001), which is similar to the standard concept of elasticity in economics.

$$CS = \frac{(ESV_j - ESV_i)/ESV_i}{(VC_{jk} - VC_{ik})/VC_{ik}} \tag{4}$$

where ESV_i and ESV_j = initial and adjusted total estimated ecosystem service values, respectively and VC_{ik} and VC_{jk} = initial and adjusted value coefficients (US\$ ha^{-1} $year^{-1}$) for LULC type 'k'. If CS is greater than one, then the estimated ecosystem value is considered elastic relative to that coefficient, but if CS is less than one, then the estimated ecosystem value is considered to be inelastic and the result will be reliable even if the value coefficient has relatively low accuracy. The greater the proportional change in the ecosystem service value relative to the proportional change in the valuation coefficient, the more critical is the use of an accurate ecosystem value coefficient (Kreuter et al., 2001; Li et al., 2010; Liu et al., 2012).

During our sensitivity studies, the value coefficients were also adjusted by consumer price index and producer price index for analyzing impact of variations of value coefficients over time using Eq. (5). The obtained average change in consumer price index was 4.9% (applied for non-market based services) and in producer price index was 5% (applied for market based services). These values were used to either discount the value coefficients which refer to 1994, if the period under consideration was before 1994, or to compound the value coefficients, if the considered period was after 1994. The latter studies allow for a direct comparison of value coefficients and changes with the gross domestic products for the periods investigated.

$$VC_{yk} = \sum (VC_{fk} \times (1 + CPI)^t \tag{5}$$

where VC_{yk} = time development value coefficient of a reference year 'y' for LULC type 'k', VC_{fk} = value coefficient of function 'f' (US\$ ha^{-1} $year^{-1}$) for LULC type 'k', CPI = average change in consumer price index in percentage applied for non-market based services while for market based services average producer price index (PPI) in percentage was used, t = number of years between 1994 and the year under consideration.

3. Results

3.1. States of estimated ecosystem service values

The ecosystem service values were estimated in the study landscape for the years 1973, 1986, 2000, and 2012 (Table 3 and Fig. 3). In general, the total ESVs of the whole study landscape were about US\$ 130.5, 118.5, 114.8 and 111.1 million in 1973, 1986, 2000 and 2012, respectively (Table 3a), when locally adapted coefficients were used. Thus, the total estimated loss of ESV was US\$ 19.3 million. The amount of ecosystem service value differed among LULC types of the entire study landscape with different reference years. For instance, at the beginning of the study period (1973), grasslands, woodlands, natural forests and water body accounted for about US\$ 12.9 million (9.9%), 11.7 million (9%), 21.4 million (16.4%), and 80.8 million (62%), respectively, of the total ecosystem service values in the study landscape (Table 3a). Croplands and tree patches shared small proportion, i.e. about US\$ 3.1 million (2.3%) and 0.6 million (0.5%), respectively, of the total ESVs. In 1986, water body also accounted for the largest part, i.e. US\$ 80.4 million (67.9%) while natural forests, grasslands, croplands and woodlands accounted for about US\$ 15.9 million (13.4%), 11.3 million (9.6%), 7 million (5.9%) and 2.1 million (1.8%), respectively, of the total ESVs in the study landscape. Tree patches and plantation forests accounted for the smallest portion of the estimated ESVs. Although the amount of estimated ESVs differed per LULC type for the year 2000 and 2012, the values showed similar order. Grasslands, croplands, and tree patches accounted for about US\$ 7.4 million (6.6%), 11.4 million (10.2%), and 1.4 million (1%), respectively, of the total ESVs in 2012. The aggregated ESVs of water body, natural forests, woodlands and plantation forests were about US\$ 91.4 million (80%) of the total value. When directly using global coefficients developed earlier, the total ESVs of the whole landscape were about US\$ 164.6, 135.8, 127.2 and 118.7 million in 1973, 1986 and 2012, respectively (Table 3b), giving a total loss of US\$ 45.9 million. This estimate is about 2.3 times higher than the loss estimated with the more conservative and better locally adapted value coefficients. The amount of ESVs also differed among LULC types depending on different reference years.

Table 3
Estimated ecosystem service values for each land use/land cover type of the different reference years in US\$ million per year of the study area using (a) own modified conservative coefficients and (b) global coefficients adopted from Costanza et al. (1997). Total ESV per year is given as a sum of each value by LULC type.

LULC types	1973		1986		2000		2012	
	ESV[a]	%	ESV[a]	%	ESV[a]	%	ESV[a]	%
a								
Bare lands	0.0	0.0	0.0	0.0	0.000	0.0	0.0	0.0
Natural forests	21.4	16.4	15.9	13.4	12.5	10.9	9.5	8.5
Plantation forests	0.0	0.0	1.1	0.9	1.7	1.5	1.3	1.1
Croplands	3.1	2.3	7.0	5.9	9.3	8.1	11.4	10.2
Grasslands	12.9	9.9	11.3	9.6	9.3	8.1	7.4	6.6
Settlements	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tree patches	0.6	0.5	0.7	0.6	0.8	0.7	1.4	1.0
Woodlands	11.7	9.0	2.1	1.8	1.0	0.9	0.7	0.6
Water	80.8	62.0	80.4	67.9	80.1	69.8	80	72.0
Total	130.5	100	118.5	100	114.8	100	111.1	100
b								
Bare lands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natural forests	43.6	26.5	32.3	23.8	25.5	20.0	19.3	16.2
Plantation forests	0.0	0.0	2.1	1.5	3.4	2.7	2.6	2.2
Croplands	1.2	0.8	2.9	2.1	3.8	3.0	4.6	3.9
Grasslands	10.7	6.5	9.4	6.9	7.8	6.1	6.1	5.2
Settlements	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tree patches	0.5	0.3	0.6	0.4	0.7	0.6	0.9	0.7
Woodlands	23.8	14.4	4.3	3.2	2.1	1.6	1.3	1.1
Water	84.8	51.5	84.3	62.1	84.1	66.1	83.9	70.7
Total	164.6	100	135.8	100	127.2	100	118.7	100

[a] ESV = ecosystem service value (million in 1994 US\$ per year).

3.2. Changes in ecosystem service values

The changes in ESVs revealed a significant reduction of the total values over the first (1973–1986), second (1986–2000), third (2000–2012) and the whole (1973–2012) study periods. The total amount of changes of ESVs during the first study period was about US \$ 12 million, which is about 9.2% of the value that existed in 1973. The total ESV was further decreased with an amount of about US\$ 3.7 million (3.1%) and 3.6 million (3.2%) of the value that existed in 1986 and 2000 during the second and third period, respectively. As mentioned above, from 1973 to 2012, the change in total ESVs was about US\$ 19.3 million, which is about 14.8% of the value that existed in 1973. The change in ESVs of each LULC type also showed a considerable reduction of the values from natural forest, woodlands and grasslands, representing about US\$ 12 (55.9%), 11 (94.5%), and 5.5 million (42.6%) of the value that existed in 1973, respectively, during the whole study period. In the same period, the values of water body also showed a slight reduction with an amount of about US\$ 0.9 million (1%) of the value that

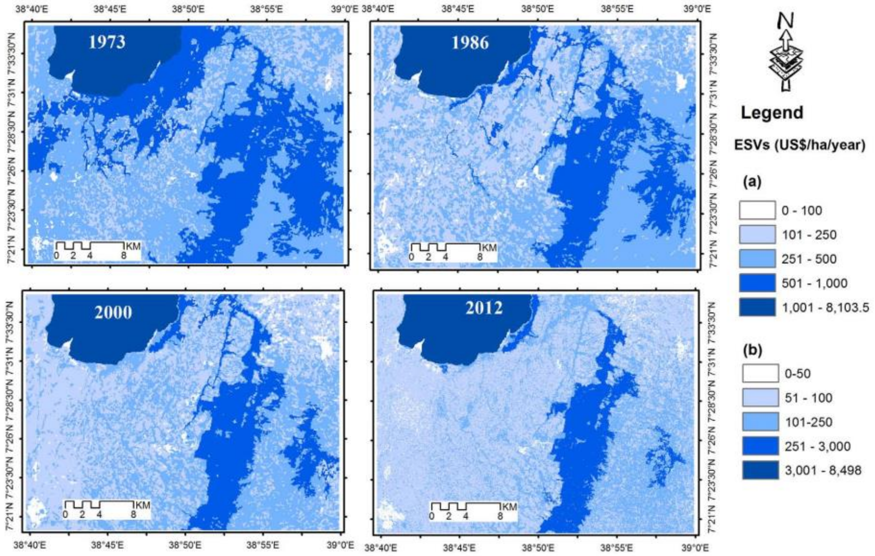

Fig. 3. Spatial distribution of ecosystem service values ($/ha/year) in the study landscape in the reference years: (a) using modified conservative coefficients and (b) global coefficients adopted from Costanza et al. (1997).

existed in 1973. On the contrary, the ESV of croplands increased during the whole study period with an amount of about US$ 8.3 million of the value that existed in 1973 (Table 4a). With the global coefficients, although the amount differed, a significant reduction of the total values was also observed over the first, second, third and the whole study period (Table 4b). Consequently, the amount of changes of ESVs also differed among LULC types of the entire landscape in different reference years. However, the percentage changes of ESV in each LULC type over the study periods are similar for both coefficients used (Table 4a & b).

3.3. Impact of land use/land cover changes on ESVs

The impact of LULC changes on ESVs notably differed among the LULC types as observed in the contributions of the area and ESV for each LULC type over the study periods (Fig. 4). In general, the area of natural forests, woodlands, grasslands and water body consistently decreased with varied proportions over the study periods. In particular, natural forests declined in size by half from about 21% in 1973 to 9% in 2012. Woodlands also declined in size from about 11.4% to 0.6% during

Table 4
Results of changes in ESVs from 1973 to 1986, 1986 to 2000, 2000 to 2012 and 1973 to 2012 time periods showing value changes in US$ million for each LULC type and percentage changes between periods in percentage (%). (a) Using own modified conservative coefficients and (b) global coefficients adopted from Costanza et al. (1997).

LULC types	Change in ecosystem service values between study periods							
	1973–1986		1986–2000		2000–2012		1973–2012	
	Million US$	Proportion (%)	Million US$	Proportion (%)	Million US$	Proportion (%)	Million US$	Proportion (%)
a								
Bare lands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natural forests	−5.6	−26.1	−3.3	−21.1	−3.1	−24.4	−12.0	−55.9
Plantation forests	1.1	100.0	0.7	67.0	−0.5	−24.8	1.3	100.0
Croplands	4.0	131.0	2.3	31.5	2.1	22.7	8.3	272.8
Grasslands	−1.5	−11.9	−2.	−17.5	−2	−21.1	−5.5	−42.6
Settlements	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tree patches	0.2	23.1	0.1	15.4	0.2	25.6	0.5	78.4
Woodlands	−9.6	−81.8	−1.2	−52.3	−0.4	−36.1	−11.0	−94.5
Water	−0.5	−0.6	−0.2	−0.3	−0.2	−0.2	−0.9	−1.1
Total	−12.0	−9.2	−3.7	−3.1	−3.6	−3.2	−19.3	−14.8
b								
Bare lands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natural forests	−11.4	−26.1	−6.8	−21.1	−6.3	−24.4	−24.4	−55.9
Plantation forests	2.1	100	1.1	67.0	−0.9	−24.8	2.6	100
Croplands	1.6	131.0	0.9	31.5	0.9	22.7	3.4	272.8
Grasslands	−1.3	−11.9	−1.7	−17.5	−1.7	−21.1	−4.6	−42.6
Settlements	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tree patches	0.1	23.1	0.1	15.4	0.2	25.6	0.4	78.4
Woodlands	−19.5	−81.8	−2.3	−52.3	−0.7	−36.1	−22.5	−94.5
Water	−0.5	−0.6	−0.3	−0.3	−0.2	−0.2	−0.9	−1.1
Total	−28.9	−17.5	−8.6	−6.3	−8.6	−6.7	−45.9	−27.9

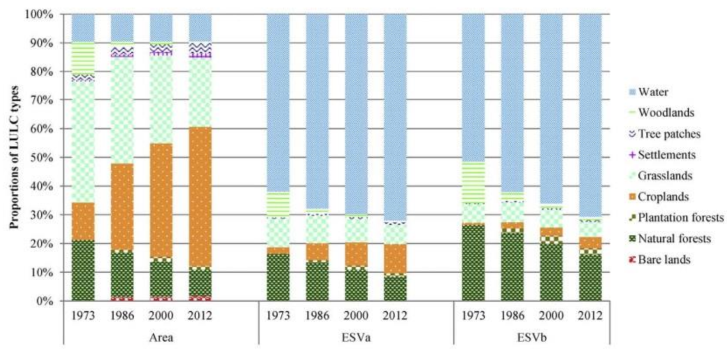

Fig. 4. Area and ecosystem services value contribution of LULC types for different reference years in the study landscape ([ESV_a = value obtained using modified conservative coefficients and ESV_b = value obtained by using the global coefficients adopted from Costanza et al. (1997)].

the same period. The area of grasslands also showed a decline from about 42.3% to 24.2%. Consequently, the total ecosystem services significantly decreased. For instance, the ESVs of natural forests decreased from 16.4% in 1973 to 8.5% in 2012. Similarly, the ESV from woodlands dropped from about 9% in 1973 to 0.6% in 2012 followed by the decline of ESVs of grasslands from 9.9% to 6.6%.

The change in the forest ecosystem (natural forests and woodlands) greatly affected the changes of the total ecosystem service values in the study area during the whole study period. For instance, in total, the ESVs of natural forests and woodlands declined by about US$ 23.8 million, while the total ESVs in the study landscape decreased by about US$ 19.3 million (Table 4a). On the other hand, the area covered by croplands significantly increased from 13 to 48.5%. However, the corresponding ESV shows only a slight increase from 2.3% in 1973 to 10.2% in 2012 compared to the area expansion of croplands. Using the global coefficients, the overall trends of ESVs as a result of LULC changes were similar (Fig. 4).

3.4. Estimated services of individual ecosystem functions and their changes

The first six individual ecosystem services based on their contributions (from high to low) to the overall value of the ecosystem services were mainly from the service category of provisioning (water supply and food production) and regulating (water regulation, water treatment, erosion control and climate regulation) for the year 1973 (Table 5a). This order of contribution changed over the different study periods. Their aggregated contribution represented about US$ 115.3, 107.8, 105.4 and 103 million in 1973, 1986, 2000 and 2012, respectively. The aggregated contribution of other ecosystem functions of each service category, i.e. provisioning (genetic resources), regulating (climate regulation, biological control, gas regulation and disturbance regulation), supporting (nutrient cycling, pollination, habitat/refugia and soil formation) and cultural services (recreation and cultural), was about US$ 15.1 million, i.e. 11.6% of the total ESVs. When comparing the values with respect to service categories of the year 1973, the results revealed

Table 5
Annual estimated value of ecosystem functions (ESV_f in US$ million US$ year^{-1}) under each service category for different reference years and their changes (1973 to 2012) in the study landscape.

Ecosystem services	Using modified conservative coefficients					Using global coefficients[a]				
	ESV_{f1973}	ESV_{f1986}	ESV_{f2000}	ESV_{f2012}	Overall change	ESV_{f1973}	ESV_{f1986}	ESV_{f2000}	ESV_{f2012}	Overall change
Provisioning services										
Water supply	21.4	21.2	21.1	21.0	−0.4	21.4	21.2	21.1	21.0	−0.4
Food production	9.4	11.7	12.7	13.6	4.2	5.1	5.3	5.2	5.2	0.0
Raw material	1.7	1.0	0.8	0.6	−1.1	10.6	6.1	4.9	3.7	−6.9
Genetic resources	1.4	0.8	0.6	0.5	−0.9	1.4	0.8	0.6	0.5	−0.9
Regulating services										
Water regulation	54.7	54.3	54.0	53.9	−0.8	54.7	54.3	54.0	53.9	−0.8
Water treatment	12.9	10.5	9.4	8.3	−4.5	13.5	11.8	10.9	10.1	−3.5
Erosion control	9.6	5.9	4.8	3.7	−5.9	9.6	5.9	4.8	3.7	−5.9
Climate regulation	7.5	4.3	3.4	2.6	−4.9	7.5	4.3	3.4	2.6	−7.0
Biological control	1.4	1.7	1.8	1.9	0.5	1.4	1.7	1.8	1.9	0.5
Gas regulation	0.8	0.6	0.5	0.4	−0.4	0.3	0.3	0.2	0.2	−0.1
Disturbance regulation	0.2	0.1	0.1	0.1	−0.1	0.2	0.1	0.1	0.1	−0.1
Supporting services										
Nutrient cycling	6.2	3.5	2.8	2.1	−4.1	30.9	17.7	14.2	10.6	−20.3
Pollination	1.6	1.6	1.6	1.5	−0.1	1.3	1.5	1.4	1.4	0.1
Soil formation	0.4	0.2	0.2	0.1	−0.2	0.4	0.2	0.2	0.1	−0.2
Habitat/refugia	0.6	0.3	0.3	0.2	−0.4	0.1	0.2	0.2	0.2	0.1
Cultural services										
Recreation	0.9	0.8	0.8	0.8	−0.1	6.1	4.5	4.1	3.6	−2.6
Cultural	0.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Sum	130.5	118.5	114.8	111.1	−19.3	164.6	135.8	127.2	118.7	−45.9

[a] Adopted from Costanza et al. (1997).

the highest value for the group of regulating services (US$ 86.9 million), followed by provisioning services (US$ 33.9 million), supporting services (US$ 8.7 million) and cultural services (US$ 1 million). This order of contribution by service categories remained the same over the different study periods. However, contribution of individual ecosystem services declined throughout the study period, except for food production (Table 5a). Although the order of contributions of ecosystem services changed in the global coefficients, the value of major contributors declined during the study period (Table 5b).

3.5. Ecosystem service sensitivity analysis

The coefficient of sensitivity (CS) was less than one in all cases indicating that the total ecosystem service values estimated in the study landscape were relatively inelastic (low sensitive) with respect to the modified conservative ecosystem value coefficients (Table 6). Adjustment to the modified value coefficient for croplands, tree patches and planation forests had very little effect on the estimate of total ESV (less than 1.17% change for a 50% change in the corresponding value coefficient). The CS ranged from a low of 0.02–0.1 for croplands, to a high of 0.62–0.72 for water body when the value coefficients for these LULC types were adjusted by 50%. The relatively low values of CS reflect the fact that the LULC area and/or the corresponding value coefficients are relatively small. CS for natural forests and water body was the highest because of the highest service value coefficient and relatively big area, which were 2nd and 5th biggest areas among the nine LULC types. Overall, the CS indicated that the ESV estimation was robust in spite of uncertainties on the modified value coefficients.

The sensitivity of ESV changes estimated by employing common conservative coefficients (million in 1994 US$ per year) and time development coefficients are shown in Table 7. By considering time variable coefficients between the study periods, the four decades change in total ESVs was about US$ 45.6 million, which is a bit more than double the loss of the value estimated using common coefficients. This additional loss is attributed to inflation effects and changes in producer prices. To gain insight for the magnitude of ESV changes in the four decades with time adjusted coefficients in comparison with GDP, the studied landscape GDP increased from about US$ 5.8 million/year in 1973 to 126.2 million/year in 2012 (Table 7). That is, the total ESV loss left with values of 2.1 times of the GDP in 2012.

4. Discussion

Datasets derived from remotely sensed imagery as a proxy measurement with their corresponding value coefficients were used to provide information regarding changes of ESVs in response to LULC dynamics in the highlands of Ethiopia, particularly focusing on the Munessa–Shashemene landscape. The study was the first of its kind in the highlands of Ethiopia that contributes to the evolving study of ecosystem service science by providing landscape level status of changes in ESVs from the region where datasets are limited. Currently, the use of proxy data is common in the estimation of ecosystem service values and their changes (Chan et al., 2006; Yoshida et al., 2010; Costanza et al., 2014; Wang et al., 2014). Quite often, such proxy data were not generated in the context of ecosystem services but are expertly re-used to map ecosystem service values due to the lack of baseline information (Maes et al., 2012). In this study, the LULC dataset produced by Kindu et al. (2013) from Landsat and RapidEye satellite images and global ecosystem service value coefficient adopted from Costanza et al. (1997) as well as own modified conservative coefficients were used to estimate total ecosystem service values and their changes.

Our approach of employing LULC datasets as a proxy of measurement facilitated the estimation process, as confirmed in previous studies (Kreuter et al., 2001; Wang et al., 2006; Li et al., 2010). The technique of calculation was proposed by Costanza et al. (1997). Afterwards, it has been applied by a number of researchers for similar studies (e.g. Kreuter et al., 2001; Zhao et al., 2004; Wang et al., 2006; Li et al., 2007; Hu et al., 2008; Kubiszewski et al., 2013). However, the quality of such valuation depends on the accuracy of LULC classification (Konarska et al., 2002) and associated value coefficients (Costanza et al., 1997; Li et al., 2010). In our study, we used spatially representative datasets of classified LULC types generated with higher overall accuracies (Kindu et al., 2013). However, the corresponding value coefficients were rough and, usually, underestimate contributions of some LULC types. The current available and widely used ecosystem service value coefficients assume spatial homogeneity of services within ecosystems (i.e. land use/land cover types). For instance, forests located either in steep slope or flat area received a similar estimation, although their ESV could greatly differ, for example concerning protection against erosion. Thus, the total ecosystem service value provided by forests would be altered depending on specific site characteristics or landscape complexity. The same is true for croplands and other LULC types found either in suitable or non-suitable area. That is, the used ecosystem value coefficients do, yet, not consider the environmental suitability of a particular land use/land cover (i.e. under the situation of LULC in the environmental land use conflict), which cause impacts to important ecosystem functions, including soil erosion, groundwater quality and habitat integrity (Pacheco et al., 2014; Valle Junior et al., 2014a, 2014b; Valle Junior et al., 2015). Such factors are also important and should be taken into account in future development of ecosystem service value coefficients to improve the understandings of the distributed characteristic of ecosystem service values. These impacts should be increasingly studied in future research initiatives.

Despite such uncertainties, as also commented by different researchers (Limburg et al., 2002; Hein et al., 2006; Maes et al., 2012), the need of accurate coefficients is often less critical for time series than for cross-sectional analyses, because value coefficients tend to affect estimates of directional change less than estimates of ecosystem values at specific points in time (Kreuter et al., 2001; Zhao et al., 2004; Tianhong et al., 2010). However, in line with considering time series and using GDP as a benchmark for comparison, it should be further investigated, how possible variation of value coefficients over time could be addressed. We updated coefficients by means of CPI and PPI, which may only be considered a very rough method. Studies about development of willingness to pay with increasing GDP are rare, so that empirical evidence for our assumption of a positive correlation between consumer and producer prices is yet not available. For the market based services (mainly agricultural products) it appears sound to

Table 6
Percentage change in estimated total ecosystem service values and coefficient sensitivity (CS) after a 50% adjustment of the modified conservative service valuation coefficients (VC).

Change of value coefficients	1973		1986		2000		2012	
	%	CS	%	CS	%	CS	%	CS
Natural forests VC ± 50%	±8.22	±0.16	±6.69	±0.13	±5.45	±0.11	±4.26	±0.09
Plantation forests VC ± 50%	±0.00	±0.00	±0.43	±0.01	±0.73	±0.01	±0.57	±0.01
Croplands VC ± 50%	±1.17	±0.02	±2.97	±0.06	±4.03	±0.08	±5.11	±0.10
Grasslands VC ± 50%	±4.93	±0.1	±4.78	±0.10	±4.07	±0.08	±3.32	±0.07
Tree patches VC ± 50%	±0.23	±0.00	±0.31	±0.01	±0.37	±0.01	±0.48	±0.01
Woodlands VC ± 50%	±4.48	±0.09	±0.90	±0.02	±0.44	±0.01	±0.29	±0.01
Water VC ± 50%	±30.98	±0.62	±33.93	±0.68	±34.91	±0.70	±35.98	±0.72

Table 7
Effect of using constant coefficients (million in 1994 US$ per year) versus time development coefficients between years to the overall estimation of ESV changes.

Approaches	Total ecosystem service values per reference year				Overall changes
	1973	1986	2000	2012	
ESV: using constant coefficients with LULC dynamics	130.45	118.45	114.78	111.14	−19.3
ESV: using time development coefficients with LULC dynamics	47.69	80.72	153.04	263.51	
ESV: using time development coefficients with assuming no LULC dynamics	47.69	88.91	173.91	309.07	
Overall changes: using LULC dynamics and time development coefficients					−45.6
GDP	5.8			126.2	

ESV = ecosystem service value (million US$ per year); time development coefficients were adjusted using consumer price index and producer price index from (http://data.imf.org/; http://faostat.fao.org/); GDP = Gross Domestic Product of the study landscape (million in 2012 US$ per year).

update value coefficients by means of PPI and also it would be intuitive to assume increased value coefficients for non-market based products when GDP and available income increases. We would suggest comparing GDP and ESV only when both are adjusted to the same year of comparison. The effect of updating value coefficients was quite strong in our study, altering the estimated change in ESV from US$ −19.3 million to US$ −45.6 million. The classical sensitivity analysis indicated that the estimated ecosystem service values for the study landscape were relatively inelastic with respect to the value coefficients, and that our estimates were relatively robust despite uncertainties in the value coefficients. Our evaluation is, thus, valid for calculating ecosystem service values over extended periods as a means of assessing their changes in response to LULC dynamics.

The quantitative results of our study with three time periods (1973–1986, 1986–2000 and 2000–2012) and whole study period from 1973 to 2012 revealed the extent of ESV changes that occurred as a result of LULC dynamics throughout the four decades. In general, the total estimated ecosystem service values declined in the study landscape throughout the study periods in both considered cases, i.e. when using the value coefficients by Costanza et al. (1997) and their modification. In particular, the ESVs of natural forests, woodlands, grasslands and the water body consistently decreased with varied proportions over the study periods. The reductions of these ecosystem service values were mainly attributed to the LULC dynamics. The conversion of natural forests, woodlands and grasslands to croplands was quite intense and common in the study landscape (Kindu et al., 2013). While our study showed that LULC dynamics in the study landscape resulted in a decline of ecosystem service values over time delivered by the affected land, it also showed that the loss depends on the interaction of changes in various LULC types. Specifically, our study suggested that the decline of ESVs were mainly linked with the huge conversion of forests, which is identified to be the main provider of ecosystem services (Martínez et al., 2009). Findings from elsewhere also showed that such changes are common in affecting the corresponding ecosystem service values. For instance, in San Antonio, Texas, Kreuter et al. (2001) estimated a cumulative loss of ESVs about US$ 6 million between 1976 and 1991. Li et al. (2007) reported a decrease of ESVs by 25.4% from 1974–2004 as a result of altered forest ecosystems in Pingbian country, China. Hu et al. (2008) also pointed out a dramatic reduction of ESV due to lose of forest areas in southwest China. A recent study on quantifying and mapping multiple ecosystem services in West Africa reported by Leh et al. (2013) also revealed a general decline of ecosystem service values from 2000 to 2009. All these studies mirror our findings that changes in LULC types have resulted in significant loss of ecosystem service values. On the other hand, increase of ESV of forests and grasslands were observed by Wang et al. (2014) in Ningxia of China, which was attributed to increase of the two LULC types. Knoke et al. (2014) also found improved ecological and economic values as a result of afforestation and intense pasturing in abandoned farmlands of the tropical Andes in southern Ecuador.

Our study further revealed that the contribution of individual ecosystem functions declined throughout the study period. Significant changes have occurred in the values of specific ecosystem service functions such as erosion control, nutrient cycling, climate regulation and water treatment, which were among the highest contributors of the total ESVs from the category of regulating and supporting services. However, the value of food production services not drastically but consistently increased during the study periods, which is mainly due to dramatic increase of croplands. Similarly, in Sanjiang Plain of northeast China, Wang et al. (2006) found changes between 1980 and 2000 favoring of a few ecosystem services (i.e. food production) at the expense of other services, such as disturbance regulation, cultural and recreation.

Our estimates of ESV changes in response to LULC dynamics have brought valuable empirical evidences that can serve for several purposes. The study provided a means to compare and highlight the magnitude of changes of ecosystem services. In this case, despite the acknowledged limitations of rough estimations, the results can be used as bases to stimulate discussions during policy formulations that can improve decisions involving various aspects of intervention strategies for sustainable use and management of land resources. It is also important to alert planners and decision makers to consider the need for revising the ongoing extensive agricultural practice at the expense of natural ecosystems. This is, especially, true for the Ethiopian highlands, where changes in LULC dynamics show a worrying trend under the current patterns of agricultural development (Hurni et al., 2005; Dessie and Kleman, 2007; Kindu et al., 2013). Quantitative evaluation of the changes in ESVs, such as this, can create common understanding of the ongoing problems serving as powerful and arguably essential communication tool to inform higher officials better with regard to trade-offs involved in land resource use options in the studied landscape as well to other area of similar settings.

5. Conclusions

In this study, detailed investigations of changes of ESVs in response to the four decades of changes in LULC types were undertaken for the first time in the Munessa–Shashemene landscape of the Ethiopian highlands. The use of GIS and datasets derived from remotely sensed imagery, with their corresponding value coefficients, as a proxy of measurement facilitated the estimation process. Our estimate revealed that LULC dynamics between 1973 and 2012 in the studied landscape, which covers about 1091 km^2 have resulted in a loss of ecosystem services ranging from US$ 19.3 million using own modified more conservative value coefficients to US$ 45.9 million using global value coefficients. The identified loss of ecosystem service values were mainly linked to the huge conversion of forests, which were the main provider of ecosystem services. The contributions of individual ecosystem functions have also declined throughout the study period. Significant changes have occurred in values of specific ecosystem service functions, such as erosion control, nutrient cycling, climate regulation and water treatment, which were among the highest contributors of the total ESVs. However, as a result of the dramatic expansion of croplands, the value of food production service function has consistently increased, though not drastically, during the study periods.

Our findings provided a database of datasets with up-to-date changes of ESVs in response to LULC dynamics for the studied landscape, which is a fast and effective way to assess the results of ecosystem service loss. Although the considered LULC datasets have the highest quality in terms of accuracy, the corresponding value coefficients were rough and our results must be considered as minimum estimate. Thus, the outputs can be used as references and bases for stimulating discussions during policy formulations that can improve decisions involving various aspects of intervention strategies for sustainable use and management of land resources. This, in turn, can lead to a management approach of ecosystems which promote gaining the maximal benefits from ecosystem services. However, to enhance the credibility of valuations for practically applicable local level policy formulation regarding alternative land resource options, it is important to obtain value coefficients for ecosystem service values that reflect local conditions. In this regard, the LULC datasets in our developed database of the studied landscape can be utilized for recalculating ecosystem service values using our approach with local level coefficients. Apart from this, evidence from the landscape level estimate indicated that the decline of ecosystem service values reflected the effects of ecological degradation to some extent. The economic gains that are aimed to be achieved under the current patterns of agricultural development in the studied landscape as well as from other area of similar setting should consider the non-market economic losses that occur as natural ecosystems (e.g. natural forests and woodlands) are lost or transformed. That means with the ongoing crop production system, a reasonable land use plan should be made with emphasis on protecting natural forests, woodlands and the water body, which have high ESVs, so as to maintain a balance between economic development and ecosystem health in the future. This can be best addressed through further studies that investigate future options and alternative strategies for sustainable use and management of land resources in the studied landscape and elsewhere with similar geographical settings.

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References

Barral, M.P., Oscar, M.N., 2012. Land-use planning based on ecosystem service assessment: a case study in the Southeast Pampas of Argentina. Agr Ecosyst Environ 154, 34–43.

Bateman, I.J., Harwood, A.R., Mace, G.M., Watson, R.T., Abson, D.J., Andrews, B., Binner, A., Crowe, A., Day, B.H., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Lovett, A.A., Munday, P., Pascual, U., Paterson, J., Perino, G., Sen, A., Siriwardena, G., van Soest, D., Termansen, M., 2013. Bringing ecosystem services into economic decision-making: land use in the United Kingdom. Science 341 (6141), 45–50.

Biazin, B., Sterk, G., 2013. Drought vulnerability drives land-use and land cover changes in the Rift Valley dry lands of Ethiopia. Agr Ecosyst Environ 164, 100–113.

Butler, J.R., Wong, G.Y., Metcalfe, D.J., Honzák, M., Pert, P.L., Rao, N., van Grieken, M.E., Lawson, T., Bruce, C., Kroon, F.J., Brodie, J.E., 2013. An analysis of trade-offs between multiple ecosystem services and stakeholders linked to land use and water quality management in the Great Barrier Reef. Australia. Agric. Ecosyst. Environ. 180, 176–191.

Central Statistical Authority (CSA), 2007. Summary and Statistical Report of the 2007 Population and Housing Census, Addis Ababa.

Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C., Daily, G.C., 2006. Conservation planning for ecosystem services. Plos Biol 4), 2138–2152.

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

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S., Turner, R.K., 2014. Changes in the global value of ecosystem services. Glob. Environ. Chang. 26, 152–158.

de Groot, R.S., Wilson, M.A., Boumans, R.M., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol. Econ. 4, 393–408.

Dessie, G., Kleman, J., 2007. Pattern and magnitude of deforestation in the South Central Rift Valley region of Ethiopia. Mt. Res. Dev. 27, 162–168.

Eshetu, Z., Högberg, P., 2000. Reconstruction of forest site history in Ethiopian highlands based on 13 C natural abundance of soils. Ambio 29, 83–89.

Frélichová, J., Vačkář, D., Pártl, A., Loučková, B., Harmáčková, Z.V., Lorencová, E., 2014. Integrated assessment of ecosystem services in the Czech Republic. Ecosyst. Serv. 8, 110–117.

Fritzsche, F., Zech, W., Guggenberger, G., 2007. Soils of the main Ethiopian Rift Valley escarpment: a transect study. Catena 70, 209–219.

Guo, Z., Xiao, X., Gan, Y., Zheng, Y., 2001. Ecosystem functions, services and their values — a case study in Xingshan County of China. Ecol. Econ. 38, 141–154.

Hein, L., van Koppen, K., de Groot, R.S., van Ierland, E.C., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecol. Econ. 57, 209–228.

Hu, H., Liu, W., Cao, M., 2008. Impact of land use and land cover changes on ecosystem services in Menglun, Xishuangbanna, Southwest China. Environ. Monit. Assess. 146, 147–156.

Hurni, H., Tato, K., Zeleke, G., 2005. The implications of changes in population, land use, and land management for surface runoff in the upper Nile basin area of Ethiopia. Mt. Res. Dev. 25, 147–154.

Kidane, Y., Stahlmann, R., Beierkuhnlein, C., 2012. Vegetation dynamics, and land use and land cover change in the Bale Mountains. Ethiopia. Environ. Monit. Assess. 184, 7473–7489.

Kindu, M., Schneider, T., Teketay, D., Knoke, T., 2013. Land use/land cover change analysis using object-based classification approach in Munessa–Shashemene landscape of the Ethiopian highlands. Remote Sens. 5, 2411–2435.

Kindu, M., Schneider, T., Teketay, D., Knoke, T., 2015. Drivers of land use/land cover changes in Munessa–Shashemene landscape of the south-central highlands of Ethiopia. Environ. Monit. Assess. 7, 452.

Knoke, T., Steinbeis, O.-E., Bösch, M., Román-Cuesta, R.M., 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.

Knoke, T., Bendix, J., Pohle, P., Hamer, U., Hildebrandt, P., Roos, K., Gerique, A., Sandoval, M.L., Breuer, L., Tischer, A., Silva, B., Calvas, B., Aguirre, N., Castro, L.M., Windhorst, D., Weber, M., Stimm, B., Günter, S., Palomeque, X., Mora, J., Mosandl, R., Beck, E., 2014. Afforestation or intense pasturing improve the ecological and economic value of abandoned tropical farmlands. Nat. Commun. 5, 5612.

Konarska, K.M., Sutton, P.C., Castellon, M., 2002. Evaluating scale dependence of ecosystem service valuation: a comparison of NOAA-AVHRR and Landsat TM datasets. Ecol. Econ. 41, 491–507.

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

Kubiszewski, I., Costanza, R., Dorji, L., Thoennes, P., Tshering, K., 2013. An initial estimate of the value of ecosystem services in Bhutan. Ecosyst. Serv. 3, e11–e21.

Leh, M.D., Matlock, M.D., Cummings, E.C., Nalley, L.L., 2013. Quantifying and mapping multiple ecosystem services change in West Africa. Agr Ecosyst Environ 165, 6–18.

Lemenih, M., Teketay, D., 2005. Effect of prior land use on the recolonization of native woody species under plantation forests in the highlands of Ethiopia. For. Ecol. Manage. 218, 60–73.

Lemenih, M., Karltun, E., Olsson, M., 2005. Assessing soil chemical and physical property responses to deforestation and subsequent cultivation in smallholders farming system in Ethiopia. Agr Ecosyst Environ 105, 373–386.

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, J., Wang, W., Hu, G., Wei, Z., 2010. Changes in ecosystem service values in Zoige Plateau. China. Agric. Ecosyst. Environ. 139, 766–770.

Limburg, K.E., O'Neill, R.V., Costanza, R., Farber, S., 2002. Complex systems and valuation. Ecol. Econ. 41, 409–420.

Liu, S., Costanza, R., Troy, A., D'Aagostino, J., Mates, W., 2010. Valuing New Jersey's ecosystem services and natural capital: a spatially explicit benefit transfer approach. Environ. Manage. 45, 1271–1285.

Liu, Y., Li, J., Zhang, H., 2012. An ecosystem service valuation of land use change in Taiyuan City. China. Ecol. Mod. 225, 127–132.

Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J.P., Grizzetti, B., Drakou, E.G., La Notte, A., Zulian, G., Bouraoui, F., Luisa Paracchini, M., Braat, L., Bidoglio, G., 2012. Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst. Serv. 1, 31–39.

Martínez, M.L., Pérez-Maqueo, O., Vázquez, G., Castillo-Campos, G., García-Franco, J., Mehltreter, K., Equihua, M., Landgrave, R., 2009. Effects of land use change on biodiversity and ecosystem services in tropical montane cloud forests of Mexico. For. Ecol. Manage. 258, 1856–1863.

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

Pacheco, F.A.L., Varandas, S.G.P., Sanches Fernandes, L.F., Valle Junior, R.F., 2014. Soil losses in rural watersheds with environmental land use conflicts. Sci. Total Environ. 485–486, 110–120.

Polasky, S., Nelson, E., Pennington, D., Johnson, K.A., 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.

Schägner, J.P., Brander, L., Maes, J., Hartje, V., 2013. Mapping ecosystem services' values: current practice and future prospects. Ecosyst. Serv. 4, 33–46.

Styers, D.M., Chappelka, A.H., Marzen, L.J., Somers, G.L., 2010. Developing a land-cover classification to select indicators of forest ecosystem health in a rapidly urbanizing landscape. Landsc.Urban Plann. 94, 158–165.

Teketay, D., Granström, A., 1995. Soil seed banks in dry Afromontane forests of Ethiopia. J. Veg. Sci. 6, 777–786.

Temesgen, H., Nyssen, J., Zenebe, A., Haregeweyn, N., Kindu, M., Lemenih, M., Haile, M., 2013. Ecological succession and land use changes in a lake retreat area (Main Ethiopian Rift Valley). J. Arid Environ. 91, 53–60.

Tianhong, L., Wenkai, L., Zhenghan, Q., 2010. Variations in ecosystem service value in response to land use changes in Shenzhen. Ecol. Econ. 69, 1427–1435.

Troy, A., Wilson, M.A., 2006. Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. Ecol. Econ. 60, 435–449.

Turner, R., Paavola, J., Cooper, P., Farber, S., Jessamy, V., Georgiou, S., 2003. Valuing nature: lessons learned and future research directions. Ecol. Econ. 46, 493–510.

Valle Junior, R.F., Varandas, S.G.P., Sanches Fernandes, L.F., Pacheco, F.A.L., 2014a. Environmental land use conflicts: a threat to soil conservation. Land Use Policy 41, 172–185.

Valle Junior, R.F., Varandas, S.G.P., Sanches Fernandes, L.F., Pacheco, F.A.L., 2014b. Groundwater quality in rural watersheds with environmental land use conflicts. Sci. Total Environ. 493, 812–827.

Valle Junior, R.F., Varandas, S.G.P., Pacheco, F.A.L., Pereira, V.R., Santos, C.F., Cortes, R.M., et al., 2015. Impacts of land use conflicts on riverine ecosystems. Land Use Policy 43, 48–62.

van der Ploeg, S., de Groot, D., 2010. The TEEB Valuation Database — A Searchable Database of 1310 Estimates of Monetary Values of Ecosystem Services, Wageningen, The Netherlands.

Wang, Z., Zhang, B., Zhang, S., Li, X., Liu, D., Song, K., Li, J., Li, F., Duan, H., 2006. Changes of land use and of ecosystem service values in Sanjiang Plain, Northeast China. Environ. Monit. Assess. 112, 69–91.

Wang, Y., Gao, J., Wang, J., Qiu, J., Bond-Lamberty, B., 2014. Value assessment of ecosystem services in nature reserves in Ningxia, China: a response to ecological restoration. PLoS One 9, e89174.

Yoshida, A., Chanhda, H., Ye, Y.-M., Liang, Y.-R., 2010. Ecosystem service values and land use change in the opium poppy cultivation region in Northern Part of Lao PDR. Acta Ecol. Sin. 30, 56–61.

Zhao, B., Kreuter, U., Li, B., Ma, Z., Chen, J., Nakagoshi, N., 2004. An ecosystem service value assessment of land-use change on Chongming Island, China. Land Use Policy 21, 139–148.

Publication IV - Manuscript

Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. (manuscript): Scenario Modelling of Land Use/Land Cover Changes in Munessa-Shashemene Landscape of the Ethiopian Highlands.

Scenario Modelling of Land Use/Land Cover Changes in Munessa-Shashemene Landscape of the Ethiopian Highlands

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Abstract

Models under sets of scenarios are used to simulate and improve our understanding of land use/land cover (LULC) changes, which is central for sustainable management of a given natural resource. In this study, we simulated and examined the possible future LULC patterns and changes in Munessa-Shashemene landscape of the Ethiopian highlands covering four decades (2012-2050) using a spatially explicit GIS-based model. Both primary and secondary sources were utilized to identify relevant explanatory variables (drivers) and LULC datasets for the model. Three alternative scenarios, namely Business As Usual (BAU), Forest Conservation and Water Protection (FCWP) and Sustainable Intensification (SI) were used. The simulated LULC map of 2012 was compared with the actual for model validation and showed a good consistency. The results revealed that areas of croplands will increase widely under the BAU scenario and would expand to the remaining woodlands, natural forests and grasslands, reflecting vulnerability of these LULC types and potential loss of associated ecosystem service values (ESVs). FCWP scenario would bring competition among other LULC types, particularly more pressure to the grassland ecosystem. Hence, the two scenarios ensure sever LULC dynamics that lead to serious environmental crisis. The SI scenario, with holistic approach, demonstrated that expansion of croplands could vigorously be reduced, remaining forests better conserved and degraded land recovered, resulting in gains of the associated total ESVs. We conclude that a holistic landscape management, i.e. SI, is the best approach to ensure expected production while safeguarding the environment of the studied landscape and elsewhere with similar geographic settings. Further study is suggested to practically test our framework through a research for development approach in a test site so that it can be used as a model area for effective use and conservation of our natural resources.

Keywords: GIS; Scenario; Regression; Datasets; Variables; Remote sensing; Model; Resource management; Ethiopia

1. Introduction

Natural resources, particularly land use/land cover (LULC) types and their associated changes were central to all related sustainable development issues (Turner, 1997) and, yet, today they are also important concerns in many regions of the world (Sleeter et al., 2013; Niedertscheider et al., 2014; Belward and Skøien, 2015; Taelman et al., 2016). It is recognized that dramatic changes in LULC can significantly modify regional climate (Fairman et al., 2011), and water balance (Davis et al., 2015), silt-up streams (Zaimes and Schultz, 2015), affect biodiversity (Dayamba et al., 2016) and ecosystems stability (Paz-Kagan et al., 2014), and disrupt socio-cultural practices (Raynaut, 2001). As pressure on LULC types increases everywhere, understanding future LULC patterns and changes is critical (Schaldach et al., 2011; Grinblat et al., 2015). These call for global attention for scientific research (Teixeira et al., 2009; Kamusoko et al., 2011; Hansen and Loveland, 2012; Belward and Skøien, 2015; Qiang and Lam, 2015).

In Ethiopia, several studies were devoted to investigate historical LULC changes and drivers behind the dynamics (Tegene, 2002; Hurni et al., 2005; Kindu et al., 2013, 2015; Temesgen et al., 2013) and their associated consequences (Daye and Healey, 2015; Kindu et al., 2016). Most of the results of these studies revealed a worrying trend of LULC changes and illustrated the complexity of the conversion process as well as the strong location dependence of the drivers in order of importance (Bewket, 2002; Kindu et al., 2015). The problem is more severe in the highlands (altitude > 1500 m) that cover nearly 44% of the total landmass of the country (Hurni et al., 2005). However, a future situations and potential consequence of this trend is not yet investigated. Development of effective land use planning strategies for sustainable resource use and conservation requires knowledge of future LULC patterns and changes (Serneels and Lambin, 2001; Bhattacharjee and Ghosh, 2015).

One way of exploring the potential future situation of land resource is through the use of modelling. Simulation models have been used by a large number of research groups to explore when and where future LULC changes would occur based on the goals of a particular study (Wu et al., 2006; Schaldach et al., 2011; Grinblat et al., 2015; Nourqolipour et al., 2015). There are differences in modelling approaches of these studies, which often relate to differences in the purpose of the study. For example, Wu et al. (2006) predicted land use change in Beijing. Grinblat et al. (2015) developed and simulated dynamics of agricultural land use in Mali, West Africa. Nourqolipour et al. (2015) employed a GIS-based model to analyze spatial and temporal development of oil palm. These results showed wide variety of information about future LULC types in each of the studied area. Since the drivers of LULC change differ from place to place, it is important to understand their location-specific interaction and reasonably predict the future demand of land, which is a key in land use planning and management specific to the areas under investigation (Wardell et al., 2003; Kindu et al., 2015; Nourqolipour et al., 2015). Otherwise, making generalization of the results to other areas might lead to erroneous conclusions.

Modelling approach to future LULC dynamics under sets of scenarios provide good information on how various courses of action may affect the future of a resource in question in which today's decision might be played out (Sun et al., 2012; Martinuzzi et al., 2015). Additionally, it gives a chance to estimate the changes of ecosystem service values (ESV) in response to LULC dynamics (Hu et al., 2008; Dallimer et al., 2015; Kindu et al., 2016). In this case, it can enrich our understanding of human activities about resource use and conservation (Bachelet et al., 2003). In particular, knowing the potential outcomes of alternative scenarios can be a powerful tool when making and implementing difficult policy decisions (Sun et al., 2012). Thus, it significantly contributes to the understanding of the potential constraints and opportunities associated with various course of actions and enhances early decision making process to minimize consequences. Therefore, the simulation outputs could play an important role in facilitating the identification and planning of management strategies that could reverse the trend and promote improved management of the resources (Bhattacharjee and Ghosh, 2015).

Although numerous spatial scenario simulation models have provided valuable insights into the LULC, particularly in the tropical region (Hall et al., 1995; Messina and Walsh, 2001; Pontius et al., 2001; Soares-Filho et al., 2006; Grinblat et al., 2015; Nourqolipour et al., 2015), future LULC patterns and changes are still poorly understood in the Ethiopian landscapes, especially in areas under extensive agricultural system (Kindu et al., 2013; Temesgen et al., 2013). In this regard, flexible spatial simulation models with the capacity to develop insights into LULC dynamics as well as explore "what if" scenarios are required. GIS can provide such extensive opportunities to model LULC dynamics in a spatially explicit manner (Nourqolipour et al., 2015; Qiang and Lam, 2015). Owning to the complex dynamics to the LULC types in a given landscape, simulating the entire landscape is also advisable for inclusive recommendations, instead of working on models that only examine a specific part of LULC types, such as modelling crop or grassland ecosystems independently (Sabatier et al., 2010; Wang et al., 2014b).

The present study, thus, takes a different and novel approach, simulating the spatial LULC patterns and changes for the entire Munessa-Shashemene landscape of the Ethiopian Highlands in the next four decades. We employed GIS-based simulation modelling approach to compose datasets mainly generated from our own research (Kindu et al., 2013, 2015, 2016) and assumptions as followed by previous studies (Messina and Walsh, 2001; Grinblat et al., 2015) under three scenarios. Thus, the approach developed for this study directly integrates major LULC types of the studied landscape, their associated drivers and assumptions for the considered scenarios. The objective was to simulate and examine the future LULC patterns and changes under each considered scenarios likely to prevail in affecting their sustainable use and conservation in the studied landscape of the Ethiopian highlands. We also estimated changes in ESVs in response to future LULC dynamics and explored opportunities and constraints under each of the considered scenario. Ultimately, findings of this study are intended to serve as early warning system for understanding the future effects of LULC dynamics and as a strategic guide to land use planning process that can better balance agricultural production and ecological conservation in the studied landscape or other areas with similar settings.

2. Materials and Methods

2.1. Study area

The study was conducted in the Munessa-Shashemene landscape of the Ethiopian highlands, which is located within Munessa and Arsi-Negele Districts (between 7°20'01.23" to 7°35'13.3"N and 38°39'43.3" to 38°59'57.31"E) about 200 km south of Addis Ababa (Figure 1). It is characterized by diverse topographic conditions with a total area of about 1,091 km^2, and it is inhabited by about 84% rural population (CSA, 2007). The elevation ranges from 1,500 m at the Central Rift Valley lakes to over 3,400 m at the Arsi-Bale massif. A mountainous and undulating topography with steep slopes characterizes the eastern part of the landscape, and gentle to moderate slopes characterizes the western part. The rainfall has a bimodal distribution. The short and main rainy seasons occur from March–May and July–September, respectively. Meteorological station records show that the annual rainfall is about 1,200 mm at Degaga town (2,000 m), which is found in the study area. Mean annual temperature is 15 °C. The soils of the area are rich in clay and classified as Mazic Vertisol in the lower altitude (1,500 m), Humic Umbrisol (2600m) and Umbric Andosol at about 3,000 m (Fritzsche et al., 2007).

The Munessa-Shashemene landscape was mainly dominated by natural forests, grasslands and woodlands in the early 1970th (Kindu et al., 2013). The natural forest belongs to the tropical dry Afromontane forest (Teketay and Granström, 1995), and the woodlands are dominated by *Acacia* species and found in the lower altitude of the study landscape. Farmers in the study landscape are engaged in agriculture with a mixed farming system, hence, crop cultivation and livestock rearing is common in all altitudinal ranges (Lemenih et al., 2005). Apart from these, the study landscape also comprises other LULC types, including, plantation forests, settlements and water bodies. The plantation forests are composed of exotic species, mainly *Cupresses lusitanica* Miller, *Pinus patula* Schlechtendal & Chamisso and *Eucalyptus* spp. During the past four decades, conversions in between LULC types of the study landscape were quit intense (Kindu et al., 2013).

<<< Insert Figure 1 here >>>>

2.2. Data used

Datasets required for this study were obtained from various sources. The LULC datasets were obtained from Kindu et al. (2013). The datasets were generated by employing object-based classification of Landsat and RapidEye satellite images with overall accuracies ranging from 85.7% to 93.2% for the different reference years. They included area statistics of nine different LULC types for the year 1973, 1986, 2000 and 2012. Six major drivers that trigger changes and influence the spatial distribution of LULC types were considered, namely population density, distance to road, distance to market, slope, rainfall and altitude (Kindu et al., 2015). The Population data were obtained from the Central Statistical and District (Woreda) Offices of the study landscape. The road network datasets were derived in ArcGIS environment from multiple sources, including Central Statistical Agency of Ethiopia, topo map, aerial photographs and RapidEye images. GPS coordinates of market centers were collected, and information regarding each center was noted. The rainfall datasets were from the WorldClim database produced by Hijmans et al. (2005) and available online. The long term annual average rainfalls (1950 to 2000) were considered for this study. The slope and altitude were derived from a 30 m digital elevation model based on Aster imagery. In addition, conservative value coefficients of the target LULC types modified by Kindu et al. (2016) was adopted to estimate the ecosystem service values and their changes in response to the future LULC dynamics of the studied landscape (Table 1).

<<< Insert Table 1 here >>>>

2.3. LULC Scenarios and future demands

Based on existing LULC related policies in Ethiopia, local demographic information and historical LULC dynamics of the studied landscape, three future scenarios have been defined to predict LULC demand for 2050, namely *Business as Usual (BAU), Forest Conservation and Water Protection (FCWP)* and *Sustainable* I*ntensification (SI)* scenarios (Table 2). The BAU scenario was designed mainly based on assumption of a continuation of LULC conversion rates of the past 40 years in the studied landscape (Kindu et al., 2013). Thus, prior to the demand calculations for 2050 in BAU scenario, the rate of change between 1973 and 2012 for each LULC types were determined using the following formula (Puyravaud, 2003):

$$r = \left(\frac{1}{(t_2 - t_1)}\right) \text{ x } \ln\left(\frac{A_2}{A_1}\right) \quad (1)$$

where r is rate of changes for each LULC type, A_1 and A_2 are the extent of each LULC type at time t_1 and t_2, respectively.

Afterwards, the trends of each LULC type during the last four decades were extrapolated using the following formula:

$$A_n = A_0 e^{rt} \quad (2)$$

where A_n is the area estimate for each LULC type in year n, A_0 is the area of base year, t is the time period that is the difference between year n and year 0, r is the average annual rate of change.

The second one, i.e., FCWP scenario, were designed for only strict implementation of spatial policies about forest conservation (FDRE, 2007) and water body protection (FDRE, 2000), in which we assume that the area of forests (natural forests, plantation and woodlands) and water body will be excluded, leading competition of the remaining LULC types to obtain the demands for LULC types in 2050. Under the SI scenario, we assume that with better family planning of projected birth rate 3% and death rate 0.85% (Garedew et al., 2012), crop productivity is likely to double with proper use of modern farm inputs (fertilizer and improved seeds), reducing average number of livestock for the households by half for improved grazing lands, restricting croplands on steep and very steep slopes (Kindu et al., 2013) through proper implementation of existing spatial policies (FDRE, 2000, FDRE, 2007), and rehabilitation of bare lands into forests to obtain the demands for LULC types in 2050. We used the projected population of 2050 with better family planning in combination with the current data on per capita settlements and croplands areas to calculate the total croplands and settlements demand for this third scenario.

<<< Insert Table 2 here >>>>

2.4. Data analyses

The spatial datasets were prepared and changed to raster format in GIS Environment at the same spatial extent with 60 m x 60 m resolution and geographical coordinates. The raster datasets were, again, converted into ASCII format in GIS Environment, and each ASCII data representing each variable was changed into a column format using MATLAB to make them suitable for statistical analysis. Afterwards, the column datasets were analyzed in SPSS using binary logistic regression to identify the relations between a set of explanatory variables (drivers) and the actual LULC patterns as dependent variables. The logistic regression was used to indicate the probability of a certain grid cell to be devoted to a LULC type given a set of driving factors (Verburg et al., 2002; Kindu et al., 2015).

$$\mathrm{Log}\left(\frac{P_i}{1-P_i}\right) = \beta_0 + \beta_1 X_{1,i} + \beta_2 X_{2,i} + \beta_3 X_{3,i} \dots + \beta_n X_{n,i} \qquad (3)$$

where Pi is the probability of a grid cell for the occurrence of the considered LULC type and the X's are the driving factors. The coefficients (β) are estimated through logistic regression using the actual land use pattern as dependent variable. The goodness of fit (i.e. performance of the logistic regression model) was evaluated using the Relative Operating Characteristic (ROC) test statistics (Lesschen et al., 2005; Kindu et al., 2015). ROC is a common measure for the goodness of fit in logistic regression to know how well the independent variables correctly predict the value of the dependent variable. A ROC value of above 0.5 indicates the spatial distribution of all of the LULC types (dependent variables) could well be explained by the selected drivers of LULC types (independent variables).

Based on the regression results, a probability map, also known as location suitability map, was produced for each LULC type. Conversion elasticity (ELAS) was specified by the user considering the actual LULC types. The conversion elasticity is an estimate of the conversion costs to account for the differences in changes between LULC types, e.g. an area with water body is not easily converted into croplands than those covered by forests. The value ranges between 0 (easy to convert) and 1 (difficult to convert). The value was defined based on user knowledge of the situation and adjusted during calibration of the model. Then, for each grid cell, the total probability was calculated for each of the LULC types based on the probability (suitability) maps from logistic model, elasticity of LULC change and the iteration variable using the formula as in Verburg et al. (2002):

$$\mathrm{TPROP}_{i,u} = \mathrm{P_{i,u}} + \mathrm{ELAS_u} + \mathrm{ITER_u} \tag{4}$$

where TROP = total probability of location i for LULC type u, $\mathrm{P_{i,u}}$ = suitability of location i for LULC type u (based on logit model), $\mathrm{ELAS_u}$ = the conversion elasticity for LULC u and $\mathrm{ITER_u}$ is an iteration variable that is specific to the land use type and indicative for the relative competitive strength of the land use type. Finally, a conversion allowance in a matrix form was specified in which a number of conversions that are not realistic were excluded, e.g., protection areas or conversion of areas with water into forests. At the end, allocation of changes in LULC was done in an iterative procedure that fulfil the demand of each scenario for the different LULC types based on the highest total probability for the considered grid cell (Figure 2).

<<< Insert Figure 2 here >>>>

Model validation was performed by generating a simulated map of 2012 LULC patterns in the studied landscape using the results of the logistic regression model, the transition matrix of the land use types and conversion elasticity. Then, the simulated LULC map was compared with the actual LULC map of 2012 derived from classification of satellite images (Kindu et al., 2013). Accordingly, overall accuracy, producer's and user's accuracies, and Kappa statistic were calculated from the error matrix. The Kappa statistic was used as one of the validation methods to evaluate the ability of the model to simulate the spatial patterns of LULC types (Pontius et al., 2001).

In order to estimate the ecosystem service values and their changes, the future LULC datasets of the considered scenarios were used as a proxy for the measurement and were prepared in the GIS environment. Accordingly, the total ESV of the landscape for each scenario, their percentage change and the values of services provided by individual ecosystem functions were estimated, respectively, using the following equation:

$$ESV = \sum(A_k \text{ x } VC_k) \quad (5)$$

$$\text{Percentage ESV change} = \left(\frac{ESV_{\text{final year}} - ESV_{\text{initial year}}}{ESV_{\text{initial year}}}\right) \times 100 \quad (6)$$

$$ESV_f = \sum(A_k \text{ x } VC_{fk}) \quad (7)$$

where ESV = total estimated ecosystem service value, A_k = the area (ha), VC_k = the value coefficient (US\$ ha^{-1} $year^{-1}$) for LULC type 'k', ESV_f = calculated ecosystem service value of function 'f', VC_{fk} = value coefficient of function 'f' (US\$ ha^{-1} $year^{-1}$) for LULC type 'k'. These resulted in summary table of the ESVs and their associated changes. The values were presented as US\$ and percentages.

3. Results

3.1. Coefficients of logistic regression

The spatial distributions of the nine LULC types were well explained by the selected drivers of LULC types, as indicated by the ROC values that measure the goodness-of-fit of the logistic regression models (Table 3). From the result, the obtained ROC values for all the LULC types were above 50%. This implies the probability distribution generated using the considered drivers of LULC types were consistent with the actual distribution of the LULC types in the studied landscape.

<<< Insert Table 3 here >>>>

3.2. Model validation

The overall accuracy for the simulated 2012 LULC map was 90.3% with the Kappa statistic of 0.889, which revealed the consistency between the simulated result and actual LULC situation (Table 4). The Kappa results show a high level of agreement for the simulated LULC map, indicating that the model is reliable for the studied landscape and can be used to simulate LULC patterns and changes under different scenarios. User's and producer's accuracies of individual classes of the simulated LULC map ranged from 72.9% (bare lands) to 92.6 (water), and 78.5 (grasslands) to 94.3% (water), respectively. This reveals that although more similarity is found between the simulated result and the actual LULC in 2012, it still exhibits a certain bias. Our producer and user accuracies of the simulated map for bare lands and grasslands (producer's accuracy) were below 79%. On the other hand, the natural forests, plantation forests, tree patches, settlements, croplands, woodlands, and water classes were relatively similar to the corresponding classes in the actual land use map for 2012.

<<< Insert Figure 3 and Table 4 here >>>>

3.3. Simulation of land use/land cover patterns under different scenarios

The simulation results of LULC distributions for the considered scenarios illustrate different patterns in the studied landscape (Figure 4). In all of the scenarios, cropland was the dominant LULC type. Under the BAU scenario (Figure 4a), the trend of expansion of croplands is clearly along the direction of sloppy areas by replacing the remaining forest areas and grasslands. Comparing the simulated scenarios for 2050 with the actual LULC map of 2012, the croplands occurred primarily towards the eastern part of the studied landscape, making up 71.9% of the study landscape followed by water (9.4%), grasslands (6.3%) and bare lands (3.7%) (Table 5). Tree patches, settlements, plantation forests, natural forests and woodlands shared small proportion of 2.8%, 2.4%, 2%, 1.9%, and 0.02%, respectively, of the entire landscape. Under the FCWP scenario, croplands also accounted for the largest part (66.3%); and water, natural forests, and grasslands accounted 9.5%, 9.2% and 5.4%, respectively, of the study landscape. Woodlands, tree patches, settlements, plantation forests, and bare lands occupied the smallest portion of the area. The results of the LULC pattern for SI scenario were similar to those of FCWP scenarios, but the area proportions were better distributed for the different LULC types. Accordingly, croplands occupied (51.1%); and grasslands, natural forests, water, tree patches accounted 16.2%, 13%, 9.5% and 4.1%, respectively, of the entire landscape. The remaining LULC types shared small portions of the study landscape.

<<< Insert Figure 4 and Table 5 here >>>

3.4. Changes in land use/land cover under different Scenarios

The change results revealed a considerable dynamics of woodlands, natural forests, grasslands, croplands and bare lands under BAU, FCWP and SI scenarios. As shown in figure 4 and table 6, under BAU scenario, croplands and bare lands increased, while natural forests, woodlands and grasslands decreased. Changes in patterns of croplands predominantly occurred within major parts of the studied landscape distributed towards the remaining natural forests, woodlands and grasslands. The total woodlands converted under BAU scenario amounted to 638.2 ha, which is about 97% of the cover that existed in 2012. Similarly, natural forest cover and grasslands decreased by 79.9 and 74%, respectively. The areas of conversion were more on steep and very steep slopes, which are more susceptible to erosion. Thus, the dramatic change of forest resources would lead to ecological degradation in the studied landscape. On the contrary, croplands and bare lands increased by 47.5 and 114 %, respectively. In addition, settlements and plantation forests had increased.

Under the FCWP scenarios, the study assumed LULC patterns and changes with only strict implementations of conservation and protection policies during the simulation years. Similar to BAU scenario, the expansion trend for croplands occurred mainly in the east, excluding forest areas of the studied landscape. It increased to a value of 18,374 ha (36.5%), which is about 10% reduction compared to the BAU scenario. Consequently, grasslands decreased from 25139 ha in 2012 to 5637.6 ha in 2050, which is about

77.6% of the cover that existed in 2012. The conversion trend of grasslands to croplands were also on steep and very steep slopes of the studied landscape, which indicated that Munessa-Shashemene still faces pressure of land degradation. However, the extent and distribution pattern of natural forests, plantation forests, woodlands and the water body were relatively similar to the conditions in 2012 of the studied landscape. The overall patterns and changes differed under SI scenario. Areas of croplands remained steady with gain of only about 5% of the cover that existed in 2012. In the same scenario, there were a considerable gain in natural forests with a value of about 3,961 ha (41.3%) and woodlands with a value of 700.3 ha (106.8%).

<<< Insert Table 6 here >>>

3.5. Ecosystem service values and their changes under different Scenarios

The total ESVs of the whole study landscape were about US$ 102.4, 109.2 and 114.1 million under BAU, FCWP and SI scenarios, respectively (Table 7). The total estimated loss of changes of ESVs under FCWP scenario was about US$ 1.9 million, which is about 1.7% of the 2012 total ESVs. The total ESV was further decreased with an amount of about US$ 8.7 million (7.8%) under BAU scenario. This estimate is about 4.6 times higher than the loss estimated under FCWP scenario. In both cases, the four decades estimated losses are less than 10% of the 2012 total ESVs, which is indicator of scarcely of ecosystems in the studied landscape. On the other hand, the SI scenario produced gains of the ESVs, which was increased with an amount of about US$ 3 million.

The estimated ESVs and their changes differed among LULC types of the entire study landscape under the considered scenarios. For instance, croplands accounted about US$ 16.7 million (16.3%) of the total ESVs, which is an increase of about US$ 5.4 million of the 2012 value under BAU scenario. The combined ESVs contribution of natural forests, woodlands and grasslands were about US$ 3.8 million (4%) of the total ESVs, while the losses were about US$ 7.6, 0.6 and 5.5 million of the 2012 value, respectively. The values of water body also showed a slight reduction with an amount of about US$ 1 million and the ESV of croplands increased with an amount of about US$ 5.4 million (Table 3). Under FCWP scenario, estimated ESVs of grasslands decreased from about US$ 7.4 to 1.7 million, while the values of croplands increased from about US$ 11.3 to 15.5 million. This loss and gain of ESVs from the two ecosystems were improved under SI scenario. The situations in forest and woodland ecosystems have also positively affected the changes of ESVs. For instance, in total, the ESVs of natural forests and woodlands improved by about US$ 4.6 million, while the total ESVs in the study landscape improved by about US$ 3 million (Table 7).

<<< Insert Table 7 here >>>

The water supply and food production of the service category of provisioning, and water regulation and water treatment of the service category of regulating were the top four service functions with the highest ESVs, contributing about US$ 94.9, 96 and 97 million under BAU, FCWP and SI scenarios, respectively (Table 8). The aggregated contribution of other thirteen ecosystem functions from each service category was about US$ 7.5 million, i.e. 7% of the total ESVs under BAU. When comparing the values with respect to service categories under BAU scenario, the results revealed the highest value for the group of regulating services (US$ 63 million), followed by provisioning services (US$ 36.6 million), supporting services (US$ 2.2 million) and cultural services (US$ 0.7 million). This order of contribution by service categories remained the same as the other scenarios. However, the amount of contribution by service categories declined under BAU and FCWP scenarios when compared with the 2012 values, except for provisioning services as a result of food production service function (Table 8). On the other hand, the contribution of individual ecosystem services showed an improvement of changes under SI in all service categories, mainly in the regulating service followed by supporting and provisioning services.

<<< Insert Table 8 here >>>

The sensitivity of ESV changes estimated by employing common conservative coefficients (million in 1994 US$ per year) and time development coefficients are also shown in Table 9. By considering time variable coefficients between the simulation periods, the total loss of ESVs during the next four decades were about US$ 124.9 million under BAU and 26.9 million under FCWP scenarios while under the SI scenario, there will be total gain of ESVs worth about US$ 43.1 million. They are more than 14 times the loss or gain of the values estimated using common coefficients, in which the higher changes of the values are attributed to inflation effects and changes in producer prices during updating value coefficients.

<<< Insert Table 9 here >>>

4. Discussion

Understanding future LULC patterns and changes is a core for science-directed sustainable management of the resources. In this study, using LULCs and their associated driver datasets, we successfully simulate the LULCs patterns and dynamics for the next four decades, mainly, by employing GIS technology in Munessa-Shashemene landscape of the Ethiopian highlands. Three kinds of scenario, i.e. BAU, FCWP, and SI scenarios were designed. Pontius et al. (2004) indicated that most spatial models contain a high level of uncertainty. Our model is validated using the actual LULC map of 2012 and showed an overall accuracy of 90.3% with a Kappa statistic of 0.889, which is within the range of accuracies found in previous similar simulation studies (Echeverria et al., 2008; Kamusoko et al., 2011). It should be noted that the simulation approach of this study only addressed the future LULC patterns and changes based on the considered scenarios, which were constructed either imperially generated datasets or assumptions as followed by previous studies (Messina and Walsh, 2001; Soares-Filho et al., 2006; Wu et al., 2006; Teixeira et al., 2009; Corner et al.; Grinblat et al., 2015; Nourqolipour et al., 2015; Qiang and Lam, 2015). They are spatially explicit, and also provide a representation of the entire landscape, i.e. modelling all major LULC types of the studied landscape. Our approach, thus, overcomes the limitations of other simulation modelling studies that examine only a specific part of LULC types, i.e. urban growth modelling (Yang and Lo, 2003), crop rotation modelling (Wang et al., 2014a) or grazing intensity modelling (Sabatier et al., 2010). However, it remains as a future scientific task, particularly comparison of the considered scenarios with mechanistic LULC modelling approach that have high potential to investigate the problem from a financial policies point of view, as suggested by Knoke et al. (2016).

Our finding on the three scenarios revealed that croplands are likely to be continued as important and influential land use type in the future sustainable use and conservation of land resources in the Ethiopian highlands. They will tend to affect patterns of LULC types differently under the considered scenarios. Specifically, under the BAU scenario, future LULC changes are likely to be more pronounced around woodlands, followed by grasslands and natural forests as a result of, mainly, by expansion of croplands. This unwanted development in the LULC system is explained in part by their spatial distribution. Woodlands occurred typically in lower elevations (below 1900 m) (Teketay, 2000) and are more of gentle to moderate slopes (Kindu et al., 2013), while natural forests tend occur in the upper elevation areas, and often more of steep to very steep slopes, making them more unsuitable for crop cultivation (Tegene, 2002; Kindu et al., 2013). Thus, future expansions of croplands will tend to overtake first suitable areas and, then, unsuitable areas of the studied landscape, making woodlands, natural forests and grasslands the most vulnerable LULC types to spatial changes. This is consistent with recent findings on extensive agricultural impacts on remaining forest resources in other tropical countries, e.g. in India (Tian et al., 2014), Zimbabwe (Baudron et al., 2011) and Tanzania (Estes et al., 2012).

Expansions of croplands also emerged as a major threat under the FCWP scenario. As a result of attempting only strict implementation of conservation and protection policies under this scenario, the pressure will be more to the grassland ecosystem that is also found on steep and very steep slopes (Kindu et al., 2013). This revealed implementation of policies targeting only specific component of LULC types (e.g. only forests or water body) will not be a solution for sustainable land resource use and conservation in the studied landscape. This may result in a failure similar to the previous efforts that have been made, where most of the remaining natural forests of the country were designated as National Forest Priority Areas (NFPAs) to stop further deforestation and forest degradation (Teketay et al., 2010). The natural forests of our studied landscape were part of the NFPAs though deforestation is still an ongoing process (Kindu et al., 2013). This problem is not limited to the landscapes of the Ethiopian highlands. Other landscapes in developing countries have also experienced similar situations. For instance, (Petursson et al., 2013) found that significant part of native forests were lost between 1973 and 1988 within the protection area of the Kenyan Mt. Elgon Forest Reserve. Similarly, (Bax et al., 2016) reported that 37% of the original forest was cleared in the Aguanytia River Basin of the central Peruvian Amazon though a national park for protection was designated within the basin.

The SI scenario showed different LULC patterns and changes compared to the other two scenarios. Cropland remained steady in terms of expansion, owning to the assumption of intensification through improved farming and better family planning under this scenario. This scenario has also given a room for recovery of forest resources due to a side-by-side implementation of existing conservation and protection policies. As a result, the landscape could be more productive and multifunctional than the current one. In this regard, our finding of this particular scenario is also consistent with the report by Estrada-Carmona et al. (2014) that suggested integrated landscape management approach in order to increase multi-functionality of landscapes for food production, livelihood improvement and ecosystem conservation. Similarly, other studies reported the role of diversified land use approach to balance food production and climate protection (Knoke et al., 2011; Schaldach et al., 2011; Knoke et al., 2013; Knoke and Hahn, 2013; Knoke et al., 2015).

Our results also revealed a decline in total ESVs under the BAU and FCWP scenarios in response to future LULC patterns and changes. The estimated loss was quadrupled under BAU compared to FCWP scenario. These reductions of ESVs were mainly linked with the huge conversion of forests, which is identified as the main provider of ecosystem services (Martínez et al., 2009; Kindu et al., 2016). Findings from elsewhere also showed that such changes are common in affecting the corresponding ecosystem service values. For instance, Hu et al. (2008) reported a dramatic reduction of ESV due to lose of forest areas in southwest China. On the other hand, our results showed improved total ESVs under SI scenario. This mirrored the findings of Wang et al. (2014) that reported an increase of total ESVs as a result of ecological restoration in Ningxia of China. Knoke et al. (2014) also found improved ecological and economic values as a result of afforestation and intense pasturing in abandoned farmlands of the tropical Andes in southern Ecuador.

The LULC patterns and changes under the considered scenarios in the studied landscape have different opportunities and constraints for resource use and conservation. As shown in the results under BAU scenario, the local people may continue with the usual way of crop cultivation for tapping the increasing cereal market opportunity of the country through exploiting the existing natural resources. For instance, expansions of croplands have been met at the expense of woodlands, natural forests and grasslands. Such kind of drastic change in LULC, however, will negatively alter the potential use of an area and may ultimately lead to loss of productivity (Tegene, 2002). This could, in turn, affect the local people by reducing the means of livelihoods of those who depend on land. Other constraint associated with the expansion of croplands at the expense of the forest resources, is the decline of biodiversity. The study landscape is known to have remnant dry Afromontane forests dominated by indigenous trees species among which some require high conservation priority (Tesfaye et al., 2010). The forest conversion over the modelled period of the BAU scenario in our study area could indirectly contribute to the loss of those high conservation priority species. Furthermore, expansion of croplands towards marginal lands on steep and very steep slopes will cause more runoff, will accelerate the soil erosion and may decrease water supplies in the absence of vegetation cover (Bewket, 2002). Such cultivation of steep and very steep slopes may result in serious degradation to the extent that the land could not sustain any crop production in the future as observed in other similar studies (Tegene, 2002; Lemenih and Teketay, 2004).

Attempting only strict implementation of conservation and protection policies under FCWP scenario will decrease the croplands by 10% compared with the BAU scenario. This could, in turn, result a deficit to feed the projected population of the studied landscape. As a consequence, the farmers living in the study landscape will be forced to migrate to the neighboring or other areas of the country. This is one of the common survival strategies of rural people in the events of food deficits due to various causes. For instance, droughts during the 1970s and early 1980s in Northern Ethiopia have caused an influx of migrants to southwestern Ethiopia (Reid et al., 2000). Tsegaye et al. (2010) also reported drought events during 1973/1974 and 1984/1985 forced influx of migrants from the neighboring Tigray to the northern Rift Valley dry lands of Ethiopia.

Under SI scenario, degraded lands could be exploited as an opportunity to catalyze restoration of native flora (Lemenih and Teketay, 2004). This is because this scenario has its largest effect in restriction of expansion of croplands and rehabilitation of degraded lands. Furthermore, the results from SI scenario, which incorporated conservations and protection policies, better family planning, and improved crop production, suggested that such integrated approaches of resource management will increase the areas of forest resources in the future, particularly on steep and very steep slopes that are not suitable for crop cultivation. Such recovery of natural habitats will increase the ecosystem services (Martínez et al., 2009; Kindu et al., 2016). Existing opportunities of incentives for conservation efforts, such as payment for ecosystem services provided through REDD+ projects (Cerbu et al., 2013) or using the area as source of tourist attraction (Juutinen et al., 2014), may also be additional source of income to the local people that can be provided by this scenario.

The holistic approach of integrated landscape management for sustainable land resource use and conservation, represented by the SI scenario, will contribute to the ongoing efforts of improving agricultural production and biodiversity conservation, which are not, as yet, simultaneously implemented in an integrated way in various part of the country (Wale, 2008; Spielman et al., 2010; Shiferaw et al., 2014). The findings are also important as our knowledge on the topic was previously restricted to historical LULC dynamics in the Ethiopian highlands (Bewket, 2002; Hurni et al., 2005; Kindu et al., 2013; Temesgen et al., 2013). It helped us to unleash the opportunities and constraints provided by this holistic approach at landscape level to ensure economic development and ecosystem health in the studied landscape or other areas with similar settings.

5. Conclusions

The study presented an important contribution to the use of spatially explicit scenario models and disclosed the ability of GIS to successfully simulate LULC patterns and changes for the first time in the Munessa-Shashemene landscape of the Ethiopian highlands. The model allowed us to make simulations for three scenarios across a range of potential social, environmental and economic policy contexts. Our simulation results indicated that croplands will expand widely in the study landscape under the BAU scenario and would expand to the remaining woodlands, natural forests and grasslands, reflecting vulnerability of these LULC types and potential loss of associated ESVs in the future. Therefore, this scenario is not suitable for sustainable resource use and conservation, highlighting the need to seriously consider such undesirable LULC developments, particularly expansion of croplands in future land use planning processes. Under the FCWP scenario, attempting only strict implementation of current protection and conservation policies will bring competition among other LULC types, particularly more pressure to the grassland ecosystem. According to the simulation outputs of this scenario, the grassland ecosystem that even occupied steep and very steep slopes will also be converted to croplands. As a result, this scenario will lead to serious environmental crisis as well. The SI scenario, having the purpose of improving the local livelihoods while ensuring healthy ecosystem in the future, showed major effect on sustainable use and conservation of resources. Adjustment of the farming system through improved crop cultivation and better family planning vigorously reduced the expansion of croplands to the remaining forest resources that are mainly found in unsuitable and sloppy areas of the studied landscape, and the rehabilitation of degraded lands facilitated to speed up vegetation recovery. These actions resulted in gaining the associated total ESVs. Consequently, the results of this scenario have implications for better landscape management by ensuring expected agricultural production while safeguarding the environment. Thus, the SI scenario confirmed the need for integrated landscape management approach, i.e. simultaneous implementation of better family planning and improved farming system to a target landscape that are critical for a sustainable future in the Ethiopian highlands beyond just conservation and protection policies.

The approach we framed in our study is a useful tool and, particularly, valuable in a situation like Ethiopia and other countries with similar conditions where there are complex processes that need to be considered for sustainable management of resources. It is flexible, spatially explicit and scenario-based, and has tremendous values for education and research applications related to future LULC change. It could also be utilized for informed decision making. The approach can also be used to identify LULC types that may deserve priority attention and enhance understanding of the potential effect of attempting only strict implementation of conservation and protection policies without considering other important issues that have consequences. Local authorities can better understand the complex LULC system using our simulated results.

In face of declining areas of the remaining forests in Munessa-Shashemene, the simulated LULC maps can also serve as an early warning system for understanding the future effects of LULC dynamics. The simulation results can also be considered as a strategic guide to land use planning process that can better balance agricultural production and ecological conservation. A further study is suggested to practically test our framework through a research for development approach in a test site, preferably, in the studied landscape or to other areas with similar settings so that it can be used as a model area for effective use and conservation of our natural resources.

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References

Bachelet, D., Neilson, R.P., Hickler, T., Drapek, R.J., Lenihan, J.M., Sykes, M.T., Smith, B., Sitch, S., Thonicke, K., 2003. Simulating past and future dynamics of natural ecosystems in the United States. Global Biogeochem. Cycles 17, n/a.

Baudron, F., Corbeels, M., Andersson, J.A., Sibanda, M., Giller, K.E., 2011. Delineating the drivers of waning wildlife habitat: The predominance of cotton farming on the fringe of protected areas in the Mid-Zambezi Valley, Zimbabwe. Biological Conservation 144, 1481–1493.

Bax, V., Francesconi, W., Quintero, M., 2016. Spatial modeling of deforestation processes in the Central Peruvian Amazon. Journal for Nature Conservation 29, 79–88.

Belward, A.S., Skøien, J.O., 2015. Who launched what, when and why; trends in global land-cover observation capacity from civilian earth observation satellites. ISPRS Journal of Photogrammetry and Remote Sensing 103, 115–128.

Bewket, W., 2002. Land Cover Dynamics Since the 1950s in Chemoga Watershed, Blue Nile Basin, Ethiopia. Mountain Research and Development 22, 263–269.

Bhattacharjee, S., Ghosh, S.K., 2015. SPATIO-TEMPORAL CHANGE MODELING OF LULC: A SEMANTIC KRIGING APPROACH. ISPRS Ann. Photogramm. Remote Sens. Spatial Inf. Sci. 2, 177–184.

Cerbu, G.A., Sonwa, D.J., Pokorny, B., 2013. Opportunities for and capacity barriers to the implementation of REDD+ projects with smallholder farmers: Case study of Awae and Akok, Centre and South Regions, Cameroon. Forest Policy and Economics 36, 60–70.

Corner, R.J., Dewan, A.M., Chakma, S. Monitoring and Prediction of Land-Use and Land-Cover (LULC) Change, 75–97.

CSA, 2007. Summary and Statistical Report of the 2007 Population and Housing Census, Addis Ababa.

Dallimer, M., Davies, Z.G., Diaz-Porras, D.F., Irvine, K.N., Maltby, L., Warren, P.H., Armsworth, P.R., Gaston, K.J., 2015. Historical influences on the current provision of multiple ecosystem services. Global Environmental Change 31, 307–317.

Davis, J., O'Grady, A.P., Dale, A., Arthington, A.H., Gell, P.A., Driver, P.D., Bond, N., Casanova, M., Finlayson, M., Watts, R.J., Capon, S.J., Nagelkerken, I., Tingley, R., Fry, B., Page, T.J., Specht, A., 2015. When trends intersect: The challenge of protecting freshwater ecosystems under multiple land use and hydrological intensification scenarios. Science of The Total Environment 534, 65–78.

Dayamba, S.D., Djoudi, H., Zida, M., Sawadogo, L., Verchot, L., 2016. Biodiversity and carbon stocks in different land use types in the Sudanian Zone of Burkina Faso, West Africa. Agriculture, Ecosystems & Environment 216, 61–72.

Daye, D.D., Healey, J.R., 2015. Impacts of land-use change on sacred forests at the landscape scale. Global Ecology and Conservation 3, 349–358.

Echeverria, C., Coomes, D.A., Hall, M., Newton, A.C., 2008. Spatially explicit models to analyze forest loss and fragmentation between 1976 and 2020 in southern Chile. Ecological Modelling 212, 439–449.

Estes, A.B., Kuemmerle, T., Kushnir, H., Radeloff, V.C., Shugart, H.H., 2012. Land-cover change and human population trends in the greater Serengeti ecosystem from 1984–2003. Biological Conservation 147, 255–263.

Estrada-Carmona, N., Hart, A.K., DeClerck, F.A., Harvey, C.A., Milder, J.C., 2014. Integrated landscape management for agriculture, rural livelihoods, and ecosystem conservation: An assessment of experience from Latin America and the Caribbean. Landscape and Urban Planning 129, 1–11.

Fairman, J.G., Nair, U.S., Christopher, S.A., Mölg, T., 2011. Land use change impacts on regional climate over Kilimanjaro. J. Geophys. Res. 116.

FDRE, 2000. Ethiopian Water Resource Management Proclamation No. 197/2000, Berhan ena Selam Printing Press, Addis Ababa, Ethiopia.

FDRE, 2007. Forest Development, Conservation and Utilization Proclamation No. 542/2007, Berhan ena Selam Printing Press, Addis Ababa, Ethiopia.

Fritzsche, F., Zech, W., Guggenberger, G., 2007. Soils of the Main Ethiopian Rift Valley escarpment: A transect study. CATENA 70, 209–219.

Garedew, E., Sandewall, M., Soderberg, U., 2012. A Dynamic Simulation Model of Land-Use, Population, and Rural Livelihoods in the Central Rift Valley of Ethiopia. Environmental Management 49, 151–162.

Grinblat, Y., Kidron, G.J., Karnieli, A., Benenson, I., 2015. Simulating land-use degradation in West Africa with the ALADYN model. Journal of Arid Environments 112, 52–63.

Hall, C.A.S., Tian, H., Qi, Y., Pontius, G., Cornell, J., 1995. Modelling Spatial and Temporal Patterns of Tropical Land Use Change. Journal of Biogeography 22, 753.

Hansen, M.C., Loveland, T.R., 2012. A review of large area monitoring of land cover change using Landsat data. Remote Sensing of Environment 122, 66–74.

Hijmans, R.J., Cameron, S.E., Parra, J.L., Jones, P.G., Jarvis, A., 2005. Very high resolution interpolated climate surfaces for global land areas. Int. J. Climatol. 25, 1965–1978.

Hu, H., Liu, W., Cao, M., 2008. Impact of land use and land cover changes on ecosystem services in Menglun, Xishuangbanna, Southwest China. Environ Monit Assess. 146, 147–156.

Hurni, H., Tato, K., Zeleke, G., 2005. The Implications of Changes in Population, Land Use, and Land Management for Surface Runoff in the Upper Nile Basin Area of Ethiopia. Mountain Research and Development 25, 147–154.

Juutinen, A., Kosenius, A.-K., Ovaskainen, V., 2014. Estimating the benefits of recreation-oriented management in state-owned commercial forests in Finland: A choice experiment. Journal of Forest Economics. 20, 396–412.

Kamusoko, C., Oono, K., Nakazawa, A., Wada, Y., Nakada, R., Hosokawa, T., Tomimura, S., Furuya, T., Iwata, A., Moriike, H., Someya, T., Yamase, T., Nasu, M., Gomi, Y., Sano, T., Isobe, T., Homsysavath, K., 2011. Spatial Simulation Modelling of Future Forest Cover Change Scenarios in Luangprabang Province, Lao PDR. Forests. 2, 707–729.

Kindu, M., Schneider, T., Teketay, D., Knoke, T., 2013. Land Use/Land Cover Change Analysis Using Object-Based Classification Approach in Munessa-Shashemene Landscape of the Ethiopian Highlands. Remote Sensing. 5, 2411–2435.

Kindu, M., Schneider, T., Teketay, D., Knoke, T., 2015. Drivers of land use/land cover changes in Munessa-Shashemene landscape of the south-central highlands of Ethiopia. Environ Monit Assess. 187.

Kindu, M., Schneider, T., Teketay, D., Knoke, T., 2016. Changes of ecosystem service values in response to land use/land cover dynamics in Munessa–Shashemene landscape of the Ethiopian highlands. Science of The Total Environment 547, 137–147.

Knoke, T., Bendix, J., Pohle, P., Hamer, U., Hildebrandt, P., Roos, K., Gerique, A., Sandoval, M.L., Breuer, L., Tischer, A., Silva, B., Calvas, B., Aguirre, N., Castro, L.M., Windhorst, D., Weber, M., Stimm, B., Günter, S., Palomeque, X., Mora, J., Mosandl, R., Beck, E., 2014. Afforestation or intense pasturing improve the ecological and economic value of abandoned tropical farmlands. Nat Comms. 5, 5612.

Knoke, T., Calvas, B., Moreno, S.O., Onyekwelu, J.C., Griess, V.C., 2013. Food production and climate protection—What abandoned lands can do to preserve natural forests. Global Environmental Change,vol. 23, 1064–1072.

Knoke, T., Hahn, A., 2013. Global Change and the Role of Forests in Future Land-Use Systems. In: Matyssek, R., Clarke, N., Cudlin, P., Mikkelsen, T., Tuovinen, J.-P., Wieser, G., Paoletti, E. (eds.), Climate Change, Air Pollution and Global Challenges. Elsevier, [S.l.], pp. 569–588.

Knoke, T., Paul, C., Härtl, F., Castro, L.M., Calvas, B., Hildebrandt, P., 2015. Optimizing agricultural land-use portfolios with scarce data—A non-stochastic model. Ecological Economics 120, 250–259.

Knoke, T., Paul, C., Hildebrandt, P., Calvas, B., Castro, L.M., Härtl, F., Döllerer, M., Hamer, U., Windhorst, D., Wiersma, Y.F., Curatola Fernández, G.F., Obermeier, W.A., Adams, J., Breuer, L., Mosandl, R., Beck, E., Weber, M., Stimm, B., Haber, W., Fürst, C., Bendix, J. 2016. Compositional diversity of rehabilitated tropical lands supports multiple ecosystem services and buffers uncertainties. Nat Comms. 7, 11877.

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

Lemenih, M., Karltun, E., Olsson, M., 2005. Assessing soil chemical and physical property responses to deforestation and subsequent cultivation in smallholders farming system in Ethiopia. Agriculture, Ecosystems & Environment 105, 373–386.

Lemenih, M., Teketay, D., 2004. Restoration of Native Forest Flora in the Degraded Highlands of Ethiopia: Constraints and Opportunities. SINET: Ethiopian Journal of Science 27, 75–90.

Lesschen, J.P., Verburg, P.H., Staal, S.J., 2005. Statistical methods for analysing the spatial dimension of changes in land use and farming systems. International Livestock Research Institute; Wageningen University, LUCC Focus 3 Office, Nairobi, Wageningen, 80 pp.

Martínez, M.L., Pérez-Maqueo, O., Vázquez, G., Castillo-Campos, G., García-Franco, J., Mehltreter, K., Equihua, M., Landgrave, R., 2009. Effects of land use change on biodiversity and ecosystem services in tropical montane cloud forests of Mexico. Forest Ecology and Management 258, 1856–1863.

Martinuzzi, S., Radeloff, V.C., Joppa, L.N., Hamilton, C.M., Helmers, D.P., Plantinga, A.J., Lewis, D.J., 2015. Scenarios of future land use change around United States' protected areas. Biological Conservation. 184, 446–455.

Messina, J.P., Walsh, S.J., 2001. 2.5D Morphogenesis: modeling land use and land cover dynamics in the Ecuadorian Amazon .Plant Ecology. 156, 75–88.

Niedertscheider, M., Kuemmerle, T., Müller, D., Erb, K.-H., 2014. Exploring the effects of drastic institutional and socio-economic changes on land system dynamics in Germany between 1883 and 2007. Global Environmental Change 28, 98–108.

Nourqolipour, R., Mohamed Shariff, A.R.B., Balasundram, S.K., Ahmad, N.B., Sood, A.M., Buyong, T., Amiri, F., 2015. A GIS-based model to analyze the spatial and temporal development of oil palm land use in Kuala Langat district, Malaysia. Environ Earth Sci. 73, 1687–1700.

Paz-Kagan, T., Shachak, M., Zaady, E., Karnieli, A., 2014. Evaluation of ecosystem responses to land-use change using soil quality and primary productivity in a semi-arid area, Israel. Agriculture, Ecosystems & Environment 193, 9–24.

Petursson, J.G., Vedeld, P., Sassen, M., 2013. An institutional analysis of deforestation processes in protected areas: The case of the transboundary Mt. Elgon, Uganda and Kenya. Forest Policy and Economics 26, 22–33.

Pontius, R., Cornell, J.D., Hall, C.A., 2001. Modeling the spatial pattern of land-use change with GEOMOD2: application and validation for Costa Rica. Agriculture, Ecosystems & Environment 85, 191–203.

Pontius, R.G., Huffaker, D., Denman, K., 2004. Useful techniques of validation for spatially explicit land-change models. Ecological Modelling 179, 445–461.

Puyravaud, J.-P., 2003. Standardizing the calculation of the annual rate of deforestation. Forest Ecology and Management 177, 593–596.

Qiang, Y., Lam, N.S.N., 2015. Modeling land use and land cover changes in a vulnerable coastal region using artificial neural networks and cellular automata. Environ Monit Assess,vol. 187.

Raynaut, C., 2001. Societies and nature in the Sahel: ecological diversity and social dynamics. Global Environmental Change 11, 9–18.

Reid, R.S., Kruska, R.L., Muthui, N., Taye, A., Wotton, S., Wilson, C.J., Mulatu, W., 2000. Landscape Ecology 15, 339–355.

Sabatier, R., Doyen, L., Tichit, M., 2010. Modelling trade-offs between livestock grazing and wader conservation in a grassland agroecosystem. Ecological Modelling 221, 1292–1300.

Schaldach, R., Alcamo, J., Koch, J., Kölking, C., Lapola, D.M., Schüngel, J., Priess, J.A., 2011. An integrated approach to modelling land-use change on continental and global scales. Environmental Modelling & Software 26, 1041–1051.

Serneels, S., Lambin, E.F., 2001. Proximate causes of land-use change in Narok District, Kenya: a spatial statistical model. Agriculture, Ecosystems & Environment 85, 65–81.

Shiferaw, B., Kassie, M., Jaleta, M., Yirga, C., 2014. Adoption of improved wheat varieties and impacts on household food security in Ethiopia. Food Policy 44, 272–284.

Sleeter, B.M., Sohl, T.L., Loveland, T.R., Auch, R.F., Acevedo, W., Drummond, M.A., Sayler, K.L., Stehman, S.V., 2013. Land-cover change in the conterminous United States from 1973 to 2000. Global Environmental Change 23, 733–748.

Soares-Filho, B.S., Nepstad, D.C., Curran, L.M., Cerqueira, G.C., Garcia, R.A., Ramos, C.A., Voll, E., McDonald, A., Lefebvre, P., Schlesinger, P., 2006. Modelling conservation in the Amazon basin. Nature 440, 520–523.

Spielman, D.J., Byerlee, D., Alemu, D., Kelemework, D., 2010. Policies to promote cereal intensification in Ethiopia: The search for appropriate public and private roles. Food Policy 35, 185–194.

Sun, X., Yue, T., Fan, Z., 2012. Scenarios of changes in the spatial pattern of land use in China. Procedia Environmental Sciences 13, 590–597.

Taelman, S.E., Schaubroeck, T., Meester, S. de, Boone, L., Dewulf, J., 2016. Accounting for land use in life cycle assessment: The value of NPP as a proxy indicator to assess land use impacts on ecosystems. Science of The Total Environment 550, 143–156.

Tegene, B., 2002. Land-Cover/Land-Use Changes in the Derekolli Catchment of the South Welo Zone of Amhara Region, Ethiopia. Eastern Africa Social Science Research Review 18, 1–20.

Teixeira, A.M.G., Soares-Filho, B.S., Freitas, S.R., Metzger, J.P., 2009. Modeling landscape dynamics in an Atlantic Rainforest region: Implications for conservation. Forest Ecology and Management 257, 1219–1230.

Teketay, D., 2000. Proceedings of the Round Table Conference on Integrated Forest Fire Management in Ethiopia. In: MOA, GTZ (eds.), Proceedings of the Round Table Conference on Integrated Forest Fire Management in Ethiopia. MoA, Addis Ababa.

Teketay, D., Granström, A., 1995. Soil seed banks in dry Afromontane forests of Ethiopia. Journal of Vegetation Science 6, 777–786.

Teketay, D., Lemenih, M., Bekele, T., Yemshaw, Y., Feleke, S., Tadesse, W., Moges, Y., Hunde, T., Nigussie, D., 2010. Forest Resources and Challenges of Sustainable Forest Management and Conservation in Ethiopia. In: Bongers, F., Tennigkeit, T. (eds.), Degraded forests in Eastern Africa. Management and restoration. Earthscan, London ;, Washington, DC, pp. 19–64.

Temesgen, H., Nyssen, J., Zenebe, A., Haregeweyn, N., Kindu, M., Lemenih, M., Haile, M., 2013. Ecological succession and land use changes in a lake retreat area (Main Ethiopian Rift Valley). Journal of Arid Environments 91, 53–60.

Tesfaye, G., Teketay, D., Fetene, M., Beck, E., 2010. Regeneration of seven indigenous tree species in a dry Afromontane forest, southern Ethiopia. Flora - Morphology, Distribution, Functional Ecology of Plants 205, 135–143.

Tian, H., Banger, K., Bo, T., Dadhwal, V.K., 2014. History of land use in India during 1880–2010: Large-scale land transformations reconstructed from satellite data and historical archives. Global and Planetary Change,vol. 121, 78–88.

Tsegaye, D., Moe, S.R., Vedeld, P., Aynekulu, E., 2010. Land-use/cover dynamics in Northern Afar rangelands, Ethiopia. Agriculture, Ecosystems & Environment 139, 174–180.

Turner, I.B., 1997. The Sustainability Principle in Global Agendas: Implications for Understanding Land-Use/Cover Change. The Geographical Journal 163, 133.

Verburg, P.H., SOEPBOER, W., VELDKAMP, A., LIMPIADA, R., ESPALDON, V., MASTURA, S.S., 2002. Modeling the Spatial Dynamics of Regional Land Use: The CLUE-S Model. Environmental Management 30, 391–405.

Wale, E., 2008. A study on financial opportunity costs of growing local varieties of sorghum in Ethiopia: Implications for on-farm conservation policy. Ecological Economics 64, 603–610.

Wang, E., Bell, M., Luo, Z., Moody, P., Probert, M.E., 2014. Modelling crop response to phosphorus inputs and phosphorus use efficiency in a crop rotation. Field Crops Research 155, 120–132.

Wang, Y., Gao, J., Wang, J., Qiu, J., Bond-Lamberty, B., 2014. Value Assessment of Ecosystem Services in Nature Reserves in Ningxia, China: A Response to Ecological Restoration. PLoS ONE 9, e89174.

Wardell, D., Reenberg, A., Tøttrup, C., 2003. Historical footprints in contemporary land use systems: forest cover changes in savannah woodlands in the Sudano-Sahelian zone. Global Environmental Change 13, 235–254.

Wu, Q., Li, H.-q., Wang, R.-s., Paulussen, J., He, Y., Wang, M., Wang, B.-h., Wang, Z., 2006. Monitoring and predicting land use change in Beijing using remote sensing and GIS. Landscape and Urban Planning 78, 322–333.

Yang, X., Lo, C.P., 2003. Modelling urban growth and landscape changes in the Atlanta metropolitan area. International Journal of Geographical Information Science 17, 463–488.

Zaimes, G.N., Schultz, R.C., 2015. Riparian land-use impacts on bank erosion and deposition of an incised stream in north-central Iowa, USA. CATENA. 125, 61–73.

Table 1. Summary land use/land cover types and assumptions for demand calculations under the three scenarios in 2050

	LULC types	Scenario 1- BAU	Scenario 2 - FCWP	Scenario 3 – SI
Assumptions	Bare lands, croplands, grasslands, natural forests, plantation forests, settlements, tree patches, water body and woodlands	Continuation of historical LULC changes	Strict implementation of spatial policies: no change allowed with in natural forests, plantation forests, woodlands and water bodies.	Strict implementation of spatial policies Change bare lands in to forests Better family planning Use improved seeds and fertilizers Restrict croplands in steep and very steep slopes

BAU = Business as usual; FCWP = Forest Conservation and Water Protection; SI = Sustainable Intensification

Table 2. Conservative value coefficients (1994 US$ ha^{-1} year^{-1}) of the studied landscape for ecosystem service functions of each LULC type adopted from Kindu et al. (2016)

	Each LULC types ecosystem service values (US$/ha/yr)						
Ecosystem services	Natural forests	Plantation forests	Croplands	Grasslands	Tree patches	Woodlands	Water
Provisioning Services							
Water supply	8	8				8	2117
Food production	32	32	187.56	117.45	117.45	32	41
Raw material	51.24	51.24				51.24	
Genetic resources	41	41				41	
Regulating Services							
Water regulation	6	6		3	3	6	5445
Water treatment	136	136		87	87	136	431.5
Erosion control	245	245		29	29	245	
Climate regulation	223	223				223	
Biological control			24	23	23		
Gas regulation	13.68	13.68		7	7	13.68	
Disturbance regulation	5	5				5	
Supporting Services							
Nutrient cycling	184.4	184.4				184.4	
Pollination	7.27	7.27	14	25	25	7.27	
Soil formation	10	10		1	1	10	
Habitat/refugia	17.3	17.3				17.3	
Cultural Services							
Recreation	4.8	4.8		0.8	0.8	4.8	69
Cultural	2	2				2	
Total	**986.69**	**986.69**	**225.56**	**293.25**	**293.25**	**986.69**	**8103.5**

Table 3. Beta (β) values for regression results of the spatial distributions of LULCs in the studied landscape.

Drivers	Land use/land cover types: Bare lands	Crop lands	Grass lands	Natural forests	Plantation forests	Settle ments	Tree patches	Water	Wood lands
Slope	-0.0231	-0.0379	-0.0044	0.0931	0.0251	-0.0472	0.0169	-0.1671	0.1610
Altitude	0.0003	-0.0021	0.0011	-0.0060	-0.0042	-0.0008	0.0002	-0.1149	-0.0671
Rainfall	-0.0005	0.0021	-0.0013	0.0239	0.0125	0.0010	0.0012	-0.0530	0.0257
Distance to road	-0.2517	-0.1089	-0.0228	0.3542	-0.1246	-0.3871	0.0521	0.5561	0.4526
Distance market	-0.0525	-0.1170	0.0254	0.0747	0.0270	-0.0592	0.0071	0.7037	-0.5407
Population density	-0.0012	-0.0010	0.0001	-0.0534	-0.0438	0.0019	0.0005	-0.1509	0.0016
Constant	-2.6525	4.8178	-4.8194	-12.5696	-5.4299	-2.1891	-5.2306	20.0617	9.1793
ROC values	0.6833	0.7443	0.6921	0.9449	0.8528	0.8473	0.6267	0.9998	0.8473

β represents the regression coefficients between LULC types and the drivers; ROC = relative operation curve; all drivers (variables) are significant at $P < 0.05$.

Table 4. Error matrix between actual land use/land cover and simulated result in 2012

Simulated LULC in 2012	Actual LULC in 2012: Bare lands	Natural forests	Plantation forests	Crop lands	Grass lands	Settle ments	Tree patches	Wood lands	Water	Total	UA (%)
Bare lands	**43**	0	0	9	2	3	0	1	1	59	72.9
Natural forests	0	**54**	3	0	0	0	3	0	0	60	90.0
Plantation forests	0	4	**54**	0	0	0	1	0	0	59	91.5
Croplands	6	0	0	**64**	7	0	0	0	0	77	83.1
Grasslands	2	2	0	5	**51**	0	2	1	0	63	81.0
Settlements	3	0	0	3		**51**	0		2	59	86.4
Tree patches	0	1	2	0	1	0	**56**	3	0	63	88.9
Woodlands	0	0	0	0	4	2	0	**44**	0	50	88.0
Water	0	0	0	0	0	1	0	3	**50**	54	92.6
Total	54	61	59	81	65	57	62	52	53	517	
PA (%)	79.6	88.5	91.5	79.0	78.5	89.5	90.3	84.6	94.3		
Overall accuracy = 90.3%; Kappa statistic = 0.889											

Where UA = user's accuracy and PA = producer's accuracy

Table 5. Summaries of area of simulated land use/land covers in 2050 under different scenarios in the studied landscape

	Actual LULC 2012		**Simulated LULC in 2050 under different scenarios**					
			BAU		**FCWP**		**SI**	
LULC types	ha	%	ha	%	ha	%	ha	%
Bare lands	1765	1.7	3792.9	3.7	3301.1	3.2	343.0	0.3
Natural forests	9588	9.24	1928.7	1.9	9588	9.2	13549	13.0
Plantation forests	1284	1.24	2073.5	2.0	1284	1.2	1284	1.2
Croplands	50317	48.5	74205.8	71.6	68691.4	66.3	52982.2	51.1
Grasslands	25139	24.2	6529.6	6.3	5637.6	5.4	16823.7	16.2
Settlements	1586	1.53	2509.9	2.4	2167.0	2.1	3187.0	3.1
Tree patches	3606	3.47	2871.9	2.8	2479.5	2.4	4279.0	4.1
Woodlands	656	0.63	17.8	0.02	656	0.6	1356.3	1.3
Water	9871	9.51	9745.5	9.4	9871	9.5	9871	9.5

BAU = Business as Usual; FCWP = Forest Conservation and Water Protection; SA = Sustainable Intensification.

Table 6. Simulated LULC changes from 2012 to 2050 showing area changed for each classes in hectare (ha) and LULC changes in percentage (%).

	Simulated changes in LULC 2012-2050 under different scenarios					
	BAU		**FCWP**		**SI**	
LULC types	ha	%	ha	%	ha	%
Bare lands	2027.9	114.9	1536.1	87	-1422.0	-80.6
Natural forests	-7659.3	-79.9	0.0	0.0	3961.4	41.3
Plantation forests	789.5	61.5	0.0	0.0	0.0	0.0
Croplands	23888.8	47.5	18374.4	36.5	2665.2	5.3
Grasslands	-18609.4	-74.0	-19501.4	-77.6	-8315.3	-33.1
Settlemts	923.9	58.3	581.0	36.6	1601.0	100.9
Tree patches	-734.1	-20.4	-1126.5	-31.2	673.0	18.7
Woodlands	-638.2	-97.3	0.0	0.0	700.3	106.8
Water	-125.5	-1.3	0.0	0.0	0.0	0.0

Table 7. Estimated ecosystem service values for each land use/ land cover type of the considered scenarios of the study landscape in US$ million per year and their changes in percentage (%) using conservative coefficients adopted from Kindu et al. (2016). Total ESV per year is given as a sum of each value by LULC type.

LULC Types	Actual -2012		BAU		FCWP		SI		Change in ESVs under considered scenarios: BAU		FCWP		SI	
	ESV	%	ESV	%	ESV	%	ESV	%	ESV	%	ESV	%	ESV	%
Bare lands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0		0.0	
Natural forests	9.5	8.5	1.9	1.9	9.5	8.7	13.4	11.7	-7.6		0.0		3.9	
Plantation forests	1.3	1.1	2.0	2.0	1.3	1.2	1.3	1.1	0.8		0.0		0.0	
Croplands	11.3	10.2	16.7	16.3	15.5	14.2	12.0	10.5	5.4		4.1		0.6	
Grasslands	7.4	6.6	1.9	1.9	1.7	1.5	4.9	4.3	-5.5		-5.7		-2.4	
Settlements	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0		0.0	
Tree patches	1.1	1.0	0.8	0.8	0.7	0.7	1.3	1.1	-0.2		-0.3		0.2	
Woodlands	0.7	0.6	0.02	0.02	0.7	0.6	1.3	1.2	-0.6		0.0		0.7	
Water body	80.0	72.0	79.0	77.1	80.0	73.2	80.0	70.1	-1.0		0.0		0.0	
Total	111.1	100	102.4	100	109.2	100	114.1	100	-8.7	-7.8	-1.9	-1.7	3.0	2.7

ESV = ecosystem service values (in million in 1994 US$ year^{-1}).

Table 8. Annual estimated value of ecosystem functions (ESV_f in US$ million US$ year^{-1}) under each service category for the considered scenarios in the study landscape using conservative coefficients adopted from Kindu et al. (2016).

	ESV_{f2012}		$ESV_{f\text{-}BAU}$		$ESV_{f\text{-}FCWP}$		$ESV_{f\text{-}SI}$	
Ecosystem Services	[a]US$	%	[a]US$	%	[a]US$	%	[a]US$	%
Provisioning Services								
Water supply	21.0	18.9	20.7	20.2	21.0	19.2	21.0	18.4
Food production	13.6	12.2	15.6	15.2	14.6	13.4	13.3	11.7
Raw material	0.6	0.5	0.2	0.2	0.6	0.5	0.8	0.7
Genetic resources	0.5	0.4	0.2	0.2	0.5	0.4	0.7	0.6
Regulating Services								
Water regulation	53.9	48.5	53.1	51.9	53.8	49.3	53.9	47.2
Water treatment	8.3	7.5	5.6	5.4	6.5	6.0	8.3	7.3
Erosion control	3.7	3.3	1.3	1.2	3.1	2.8	4.6	4.0
Climate regulation	2.6	2.3	0.9	0.9	2.6	2.4	3.6	3.2
Biological control	1.9	1.7	2.0	1.9	1.8	1.7	1.8	1.5
Gas regulation	0.4	0.3	0.1	0.1	0.2	0.2	0.4	0.3
Disturbance regulation	0.1	0.1	0.0	0.0	0.1	0.1	0.1	0.1
Supporting Services								
Nutrient cycling	2.1	1.9	0.7	0.7	2.1	1.9	3.0	2.6
Pollination	1.5	1.4	1.3	1.3	1.2	1.1	1.4	1.2
Soil formation	0.1	0.1	0.0	0.0	0.1	0.1	0.2	0.2
Habitat/refugia	0.2	0.2	0.1	0.1	0.2	0.2	0.3	0.2
Cultural Services								
Recreation	0.8	0.7	0.7	0.68	0.7	0.7	0.8	0.7
Cultural	0.0	0.0	0.0	0.01	0.0	0.0	0.0	0.0
Sum	111.1	100	102.4	100	109.2	100	114.1	100

Table 9: Effect of using constant coefficients (million in 1994 US$ per year) versus time development coefficients to the simulated years to the overall estimation of ESV changes (adopted from Kindu et al., 2016).

	Total ecosystem service values for actual and simulated years				Overall changes: using constant coefficients with LULC dynamics		
Approaches	2012	BAU	FCWP	SI	BAU	FCWP	SI
ESV: using constant coefficients with LULC dynamics	111.14	102.43	109.24	114.10	-8.7	-1.9	3.0
ESV: using time development coefficients with LULC dynamics	263.51	1453.62	1551.65	1621.69			
ESV: using time development coefficients with assuming no LULC dynamics	263.51	1578.58	1578.58	1578.58			
Overall changes: using dynamics and time development coefficients	0.00	-124.96	-26.93	43.11			

ESV = Ecosystem service value (million US$ per year); time development coefficients were adjusted using consumer price index and producer price index

Captions for figures

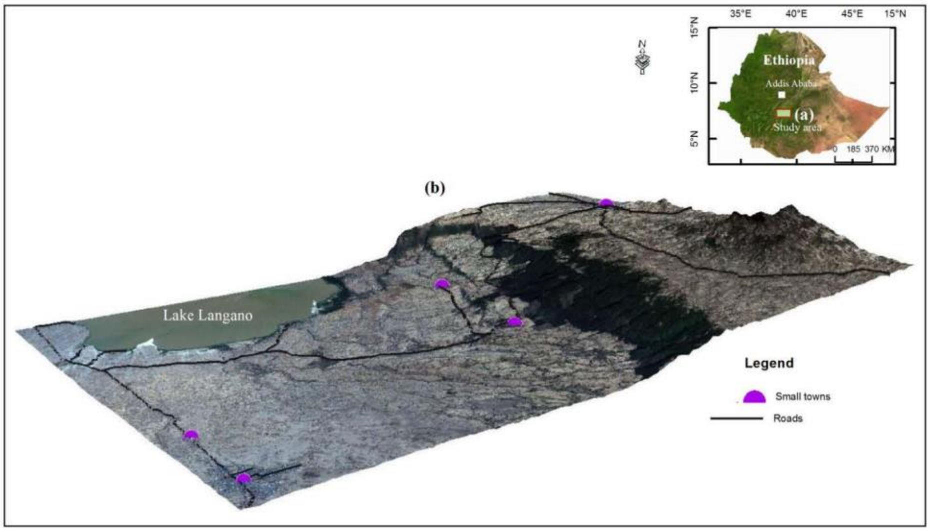
Figure 1. Location of the study area: (**a**) Ethiopian context, (**b**) 3D view of study landscape. Background is RapidEye image of year 2012 with RGB (red, green, blue): bands 3, 2 and 1, draped on Aster Digital Elevation Model (DEM) with three-fold vertical exaggeration.

Figure 2. Methodological approach to investigate future LULC patterns and changes using scenario modelling

Figure 3. Actual (**a**) and simulated (**b**) land use/land cover of the year 2012 in the study landscape

Figure 4. Land use/land cover pattern in 2050 under different scenarios in the study landscape (**a**) Business as Usual scenario, (**b**) Forest Conservation and Water Protection scenario and (**c**) Sustainable Intensification scenario

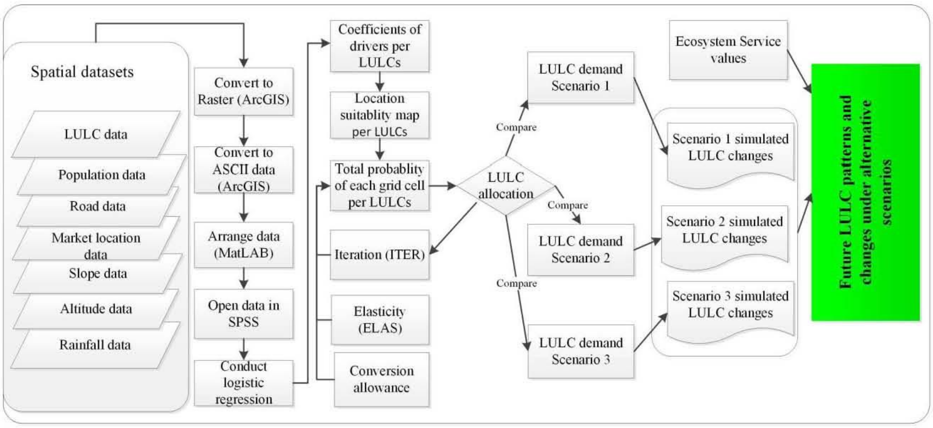
Figure 1


35°E
40°E
45°E
15°N
10°N
5°N
Ethiopia
Addis Ababa
(a)
Study area
0 185 370 KM
(b)
Lake Langano
Legend
Small towns
Roads

29

741 Figure 2

742



743

744

30

Figure 3

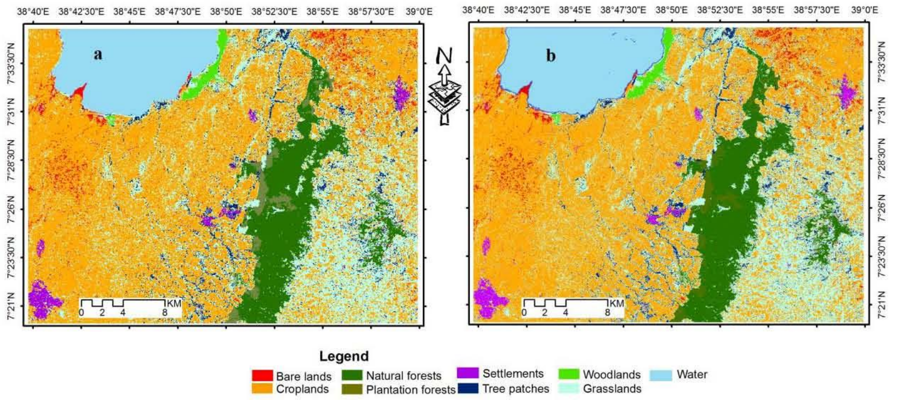
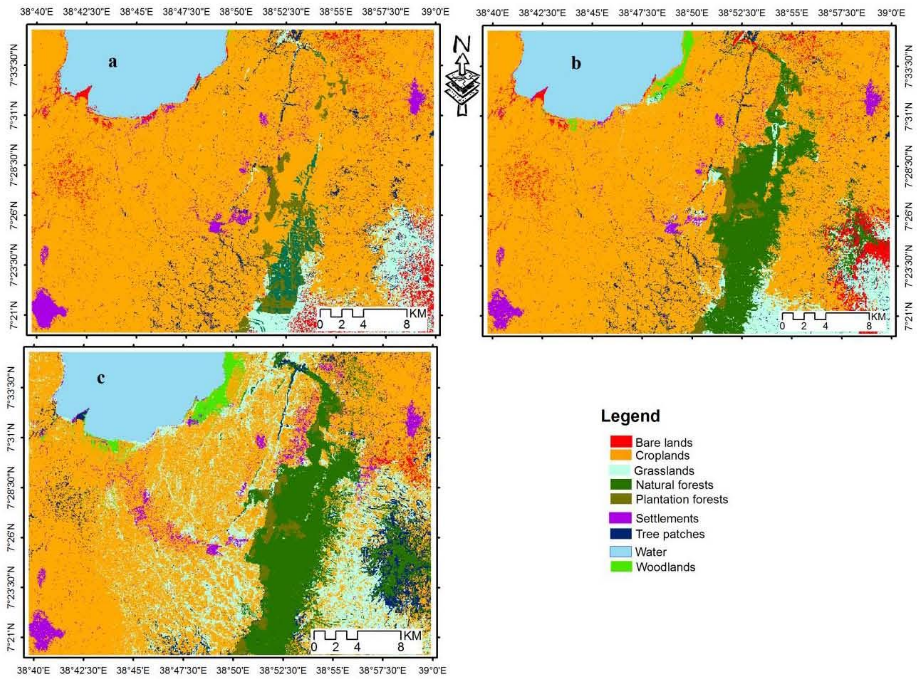

a
b
N
38°40'E
38°42'30"E
38°45'E
38°47'30"E
38°50'E
38°52'30"E
38°55'E
38°57'30"E
39°0'E
7°33'30"N
7°31'N
7°28'30"N
7°26'N
7°23'30"N
7°21'N
0 2 4 8 KM
Legend
Bare lands
Croplands
Natural forests
Plantation forests
Settlements
Tree patches
Woodlands
Grasslands
Water

Figure 4


a
b
c
N
0 2 4 8 KM
38°40'E 38°42'30"E 38°45'E 38°47'30"E 38°50'E 38°52'30"E 38°55'E 38°57'30"E 39°0'E
7°21'N 7°23'30"N 7°26'N 7°28'30"N 7°31'N 7°33'30"N
Legend
Bare lands
Croplands
Grasslands
Natural forests
Plantation forests
Settlements
Tree patches
Water
Woodlands

List of all publications by the author

The list below (ordered descending from newer to older publication dates) only includes those contributions during this PhD research project period of which 12 are from the project.

Scientific journals (peer reviewed):

1. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. Scenario Modelling of Land Use/Land Cover Changes in Munessa-Shashemene Landscape of the Ethiopian Highlands. (manuscript in final stage of prep. to be submitted)
2. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2016. Changes of Ecosystem Service Values in Response to Land Use/Land Cover Dynamics in Munessa-Shashemene Landscape of the Ethiopian Highlands. Science of the Total Environment 547:137-147.
3. Adhikari, R.K.; **Kindu, M.**; Pokharel, R.; Castro, L.M.; Knoke, T. 2016. Financial compensation for biodiversity conservation in Ba Be National Park of Northern Vietnam. Journal for Nature Conservation 35: 92–100. Doi: 10.1016/j.jnc.2016.12.003.
4. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2015. Drivers of land Use/Land Cover Changes in Munessa-Shashemene Landscape of the south-central highlands of Ethiopia. Environmental Monitoring and Assessment 187(7). Doi: 10.1007/s10661-015-4671-7.
5. Desalegn, T.; Cruz, F.; **Kindu, M.**; Turrión, M.; Gonzalo, J. 2014. Land-use/land-cover (LULC) change and socioeconomic conditions of local community in the central highlands of Ethiopia. International Journal of Sustainable Development & World Ecology 21 (5), 406–413. Doi: 10.1080/13504509.2014.961181.
6. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2013. Land Use/Land Cover Change Analysis Using Object-Based Classification Approach in Munessa-Shashemene Landscape of the Ethiopian Highlands. Remote Sensing, 5(5): 2411-2435.
7. Temesgen, H.; Nyssen, J.; Zenebe, A.; Haregewey, N.; **Kindu, M.**; Lemenih, M.; Haile, M. 2013. Ecological succession and land use changes in a lake retreat area (Main Ethiopian Rift Valley). Journal of Arid Environments. 91. P.53-60.

Other journals (non-reviewed):

1. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2016. Landnutzungs-Änderungen im Hochland von Äthiopien. Allgemeine Forst Zeitschrift für Waldwirtschaft und Umweltvorsorge 71 (13): 27-30
2. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2014. Änderungen der Landnutzungim Hochland von Äthiopien: Erfasst mithilfe von GIS und Fernerkundung, Allgemeine Forst Zeitschrift für Waldwirtschaft und Umweltvorsorge (AFZ, Der Wald), 2014, 21: 34-36.

3. Kindu, M.; Schneider, T.; Knoke, T. 2013. Monetäre Bewertung der Landnutzungs-/Landoberflächentypen im Landschaftsausschnitt Munessa-Shashemene des äthiopischen Hochlandes, Allgemeine Forst Zeitschrift für Waldwirtschaft und Umweltvorsorge (AFZ, Der Wald), 2013, 68 (21): 12-14.

Book chapter:

1. Schneider, T.; Elatawneh, A.; Rahlf, J.; **Kindu, M.**; Rappl, A.; Thiele, A.; Boldt, M.; Hinz ,S. 2013. Parameter Determination by RapidEye and TerraSAR-X Data: A Step Toward a Remote Sensing Based Inventory, Monitoring and Fast Reaction System on Forest Enterprise Level; In: Krisp JM, Meng L, Pail R, Stilla U, eds. 2013. Lecture Notes in Geoinformation and Cartography. Berlin, Heidelberg: Springer Berlin Heidelberg, S. 81-107.

Conference proceedings and abstracts:

1. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2016. GIS based Scenario Modelling of Land Use/Land Cover Changes in Ethiopia; Environmental System Research Institute (ESRI) International Conference, June 27- July 1, 2016, San Diego, California, USA.
2. Wallner, A.; Staub, C.; **Kindu, M.**; Tian, J.; Schneider, T. 2016. Supporting forest inventories via satellite derived height and spectral data; European Association of Remote Sensing Laboratories (EARSeL), 3rd Workshop of the Special Interest Group on Forestry, 15-16 September 2016, Krakow, Poland.
3. Schneider, T.; Tian, J.; **Kindu, M.**; Knoke, T. 2015. Optionen der Auswertung multispek-tral-stereoskopischer Daten im Forst-bereich am Beispiel eines Ziyuan-3 Datensatzes; Der gepixelte Wald – Reloaded" am Zentrum Wald-Forst-Holz Weihenstephan, März 2015, Freising/München.
4. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2014. GIS based analysis of causes for land – use/cover change in Ethiopia; 2014 Environmental System Research Institute (ESRI) International Conference, July 14-48, 2014, San Diego, California, USA.
5. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2014. Assessment of Drivers behind Loss of Forests in Munessa-Shashemene Landscape of the Ethiopian Highlands; International Union of Forest Research Organization (IUFRO) Forest Change Conference, April 2014, Freising/München.
6. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. 2014. Classifying Land Use/Land Covers in Munessa-Shashemene Landscape of the Ethiopian Highlands Using Mono-Temporal RapidEye Image, In: Mosandl R., Weber M., Stimm B., Hildebrandt P., Knoke T. (eds.): Tropical Ecosystems – between Protection and Production – Conference of the Society for Tropical Ecology, Freising. ISBN 978-3-00-045147-8.

7. Corti, N.; **Kindu, M**.; Schneider, T.; Knoke, T. 2014. Forest Burned Area Mapping in Southern Ecuador using mono-temporal Rapid Eye imagery and Object Based Image Analysis, In: Mosandl R., Weber M., Stimm B., Hildebrandt P., Knoke T. (eds.): Tropical Ecosystems – between Protection and Production – Conference of the Society for Tropical Ecology, Freising. ISBN 978-3-00-045147-8.
8. Kindu, M.; Elatawneh, A.; Corti, N.; Wallner, A.; Felbermeier, B.; Cabra, R.A.; Schneider, T.; Knoke, T. 2013. Evaluating Developed Rule Sets Transferability for Extracting Forest Resources from RapidEye Data“; Living Planet Symposium for the European Space Agency (ESA) in Edinburgh, United Kingdom.
9. Kindu, M.; Schneider, T.; Knoke,T.; Teketay,D.; Tibebe, D. 2012. Multi-temporal analysis of land use/land cover dynamics using satellite images in natural coffee regions of south west Ethiopia. European Association of Remote Sensing Laboratories (EARSeL) 1st Workshop on Temporal Analysis of Satellite Images from May 23-25, 2012, Myknos, Greece.
10. Kelemu, K.; **Kindu, M.**; Yetneberk, S. 2012. Establishing Value Chain Model to Commercialize Moringa Production In: Tadesse W., Desalegn G., Yirgu A. (eds.): Forestry and Forest Products in Ethiopia – Proceedings of The National Workshop on Forestry Research Technologies Dissemination, Addis Ababa, Ethiopia. ISBN 978-99944-53-869-3.
11. Knoke, T.; **Kindu, M.** 2011. Montane tropical forests as part of economically diversified land-use portfolios. Proceedings of Association for Tropical Biology and Conservation (ATBC) 2011 conference, Arusha, Tanzania.