Science of the Total Environment 675 (2019) 354–366

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Implications of land use/land cover dynamics and *Prosopis* invasion on ecosystem service values in Afar Region, Ethiopia

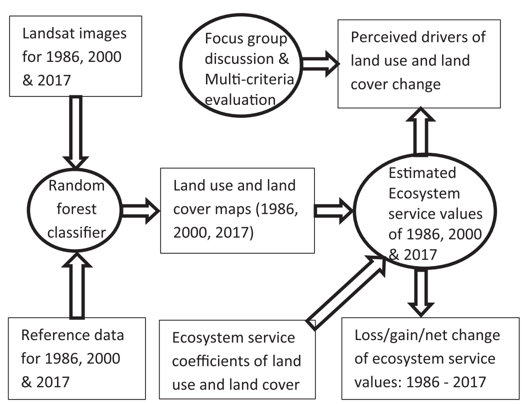
Hailu Shiferaw [a,b,*], Woldeamlak Bewket [b], Tena Alamirew [a], Gete Zeleke [a], Demel Teketay [c], Ketema Bekele [d], Urs Schaffner [e], Sandra Eckert [f]

[a] Water and Land Resource Centre of Addis Ababa University, P.O. Box 3880, Addis Ababa, Ethiopia
[b] Department of Geography and Environmental Studies of Addis Ababa University, P.O. 1176, Addis Ababa, Ethiopia
[c] Botswana University of Agriculture and Natural Resources (BUAN), Department of Crop Science and Production, Private Bag 0027, Gaborone, Botswana
[d] School of Agricultural Economics & Agribusiness of Haramaya University, P.O. Box 50, Dire Dawa, Ethiopia
[e] CABI, Rue des Grillons 1, CH-2800 Delémont, Switzerland
[f] Centre for Development and Environment, University of Bern, Mittelstrasse 43, CH-3012 Bern, Switzerland

HIGHLIGHTS

- LULC transformation analysis (1986-2017) showed that grassland and bush-shrub-land losing fast to *Prosopis*.
- *Prosopis* invasion increased at annual rates of 31,127 ha while grassland and bush-shrub-woodland declined with 19,312 ha and 10,543 ha, respectively.
- Local communities perceive that climate change, frequent droughts and invasive species as the major drivers of LULC changes.
- The ESVs loss between the two periods is estimated at US$ 602 million (ranged 112 to 1,092 million) within 31 years.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 25 January 2019
Received in revised form 3 April 2019
Accepted 13 April 2019
Available online 17 April 2019

Editor: Ralf Ludwig

Keywords:
Ecosystem service values
Land use and cover dynamics
Machine learning algorithm
Random forest classifier
Prosopis juliflora
Ethiopia

ABSTRACT

Land use/land cover (LULC) dynamics and the resulting changes in ecosystems, as well as the services they provide, are a consequence of human activities and environmental drivers, such as invasive alien plant species. This study assessed the changes in LULC and ecosystem service values (ESVs) in the Afar National Regional State, Ethiopia, which experiences a rapid invasion by the alien tree *Prosopis juliflora* (Swartz DC). Landsat satellite data of 1986, 2000 and 2017 were used in Random Forest algorithm to assess LULC changes in the last 31 years, to calculate net changes for different LULC types and the associated changes in ESVs. Kappa accuracies of 88% and higher were obtained for the three LULC classifications. Post-classification change analyses for the period between 1986 and 2017 revealed a positive net change for *Prosopis* invaded areas, cropland, salt flats, settlements and waterbodies. The rate of *Prosopis* invasion was estimated at 31,127 ha per year. Negative net changes were found for grassland, bareland, bush-shrub-woodland, and natural forests. According to the local community representatives, the four most important drivers of LULC dynamics were climate change, frequent droughts, invasive species and weak traditional law. Based on two different ESVs estimations, the ecosystem changes caused by LULC changes resulted in an average loss of ESVs in the study area of about US$ 602 million (range US$ 112 to

* Corresponding author at: Private P.O. B.7985, A.A., Ethiopia.
E-mail addresses: Hailu2nd@gmail.com, Hailushi31@yahoo.com (H. Shiferaw).

https://doi.org/10.1016/j.scitotenv.2019.04.220

1091 million) over the last 31 years. With an increase in area by 965,000 ha, *Prosopis*-invaded land was the highest net change during the study period, followed by grassland (−599,000 ha), bareland (−329,000 ha) and bush-shrub-woodland (−327,000 ha). Our study provides evidence that LULC changes in the Afar Region have led to a significant loss in ESVs, with serious consequences for the livelihoods of the rural people.

1. Introduction

Land use/land cover (LULC) changes are aspects of global environmental change and affect ecosystem processes and services. For example, an increasing demand for agricultural, industrial or urban areas compromises the ability of natural forests, waterbodies and grasslands to support mankind (Nelson et al., 2009; Goldman-Benner et al., 2012). In recent decades, a large amount of change in LULC has been observed, caused by different socio-economic and biophysical drivers, such as population growth, agricultural expansion and intensification, accessibility to infrastructure and markets, water availability or climate. An additional driver of global change is invasive alien plant species (IAPS), which causes significant impacts on biodiversity and ecosystem services (ES) (Vilà et al., 2011; Vilà and Hulme, 2017), or promote ecosystem disservices (EDS) (Shackleton et al., 2016; Vaz et al., 2017), and thus alter the benefits people derive from nature (Pejchar and Mooney, 2009). However, little is known about the effect of IAPS on LULC dynamics and its consequences of ecosystems services at larger spatial scales (Le Maitre et al., 2014).

Species-rich ecosystems are able to simultaneously provide multiple ES (Lefcheck et al., 2015). If LULC changes negatively affect biodiversity and the provisioning of these ES, or promote EDS, they also reduce the overall value of the land. According to TEEB (2010), "recognizing value in ecosystems, landscapes, species and other aspects of biodiversity is a feature of all human societies and communities, and sometimes sufficient to ensure conservation and sustainable use". Over the last 20 years, many ecosystem service values (ESVs) studies have been carried out at global, national or subnational levels (Schmidt et al., 2016), some of which integrating spatially explicit approaches (Kremer et al., 2016; Liu et al., 2009). At the global level, the value of ES in 2011 was estimated at US$ 125 trillion per year (assuming changes in LULC) to US$ 145 trillion per year (without assuming LULC changes), and the decrease in ESVs between 1997 and 2011 as a result of land use changes was at US$ 4.3 to 20.2 trillion per year (Costanza et al., 2014). Losses in ESVs at national and subnational levels were also reported to be high (e.g. Crespin and Simonetti, 2016; Sutton et al., 2016). Quantification of ESVs based on the ES database (Van der Ploeg et al., 2010) is commonly undertaken by integrating LULC data of biomes present in a region of interest (Costanza et al., 2014; Van der Ploeg et al., 2010). Although these biomes are not exactly similar in their characteristics and functions with the LULC types used in different studies, average values per unit area derived from valuation studies for a particular biome can be used as proxies for estimating the ESVs of the corresponding LULC types (Tolessa et al., 2017).

Braat and Groot (2012) suggested that decisions regarding the future use of an ecosystem should consider the full costs and benefits for the welfare of the current and future generations. ES are the products of complex interconnected social–ecological systems (Grêt-Regamey et al., 2015), indicating that evaluating ES supply and values requires a deep understanding of the social–ecological systems and the dynamics of the relationship between human activities and the ecosystems they rely on (Grêt-Regamey et al., 2015; Maes et al., 2012; Shackleton et al., 2016; Vaz et al., 2017). Hence, understanding ES and their values as well as their spatial dynamics will contribute towards considering ES for policy goals and measuring welfare of society both at national and subnational levels (Niquisse et al., 2017).

In sub-Saharan Africa, some studies have been conducted on mapping and valuation of ES in the context of LULC changes (Arowolo et al., 2018; Hulme et al., 2013; Leh et al., 2013; Kindu et al., 2016; Tolessa et al., 2017; Silvestri et al., 2013). Almost all studies indicate that this region is under severe pressure of degradation, with significant consequences for rural livelihoods (Scholes et al., 2018). For example, Sutton et al. (2016) estimated for Ethiopia a loss of 17.7% in ESVs due to land degradation, which is also reflected in studies conducted in different parts of Ethiopia (Gashaw et al., 2018; Kindu et al., 2016; Tolessa et al., 2017). Drivers of land degradation in sub-Saharan Africa include the expansion of crop production, unsustainable grazing and forestry practices, and climate change (Scholes et al., 2018). The impact of invasive species on losses in ESVs in sub-Saharan Africa may be particularly relevant in this region, as low-income countries appear to be particularly vulnerable to biological invasions in relative terms (potential economic impact as a proportion of gross domestic product; Paini et al., 2016).

The South American tree, *Prosopis juliflora* (Swartz DC.), hereafter referred to as *Prosopis*, is one of the world's worst invasive species (Lowe et al., 2000). *Prosopis* was introduced to different parts of the world with the aim of providing benefits to rural people, such as the production of fuelwood, charcoal and construction material, as well as to stabilize soil in degraded ecosystems (Shackleton et al., 2014). However, *Prosopis* has become invasive in many places and is increasingly known for its negative ecological and socio-economic impacts (Keller et al., 2010; Shackleton et al., 2015a; Shiferaw et al., 2004). With regard to the *Prosopis* invasion and LULC changes, some studies have been carried out in Ethiopia at Kebele (the lowest administrative unit in Ethiopia; Ayanu et al., 2014) and District/Woreda level (Engda, 2009; Haregeweyn et al., 2013). However, LULC change analysis and estimation of ESVs in the context of *Prosopis* invasion has not been assessed in any parts of the invaded area so far.

This study aimed at assessing and analyzing LULC changes in the Afar National Regional State (ANRS), Ethiopia, and at quantifying the contribution of *Prosopis* invasion to changes in LULC and its consequences on ESVs. Landsat satellite data of three points in time, 1986, 2000 and 2017 were used to quantify changes in LULC over the last 31 years. Moreover, associated drivers of the LULC dynamics were identified in focus group discussions with local stakeholders. The specific objectives were to: i) assess LULC dynamics, calculate its gains, losses and net changes in area of the different LULC types, and quantify *Prosopis* invasion rates over the last 31 years, ii) identify the local stakeholders' perceptions of the drivers of LULC change, and iii) estimate the ESV changes caused by LULC dynamics in the study area in order to assist policy makers in designing evidence-based solutions.

2. Materials and methods

2.1. Study area and Prosopis invasion

The study area consisted of the southern part of the ANRS of Ethiopia and comprised 6.67 million ha (Fig. 1). The northern part of the region was excluded as it is mainly occupied by active volcanic materials and sandy and/or rocky areas and has hardly been invaded by *Prosopis* so far. The ANRS has a long term mean annual rainfall of about 560 mm per year (average for the period 1965–2002: Shiferaw et al., 2004). The region is the hottest part of Ethiopia with a mean annual

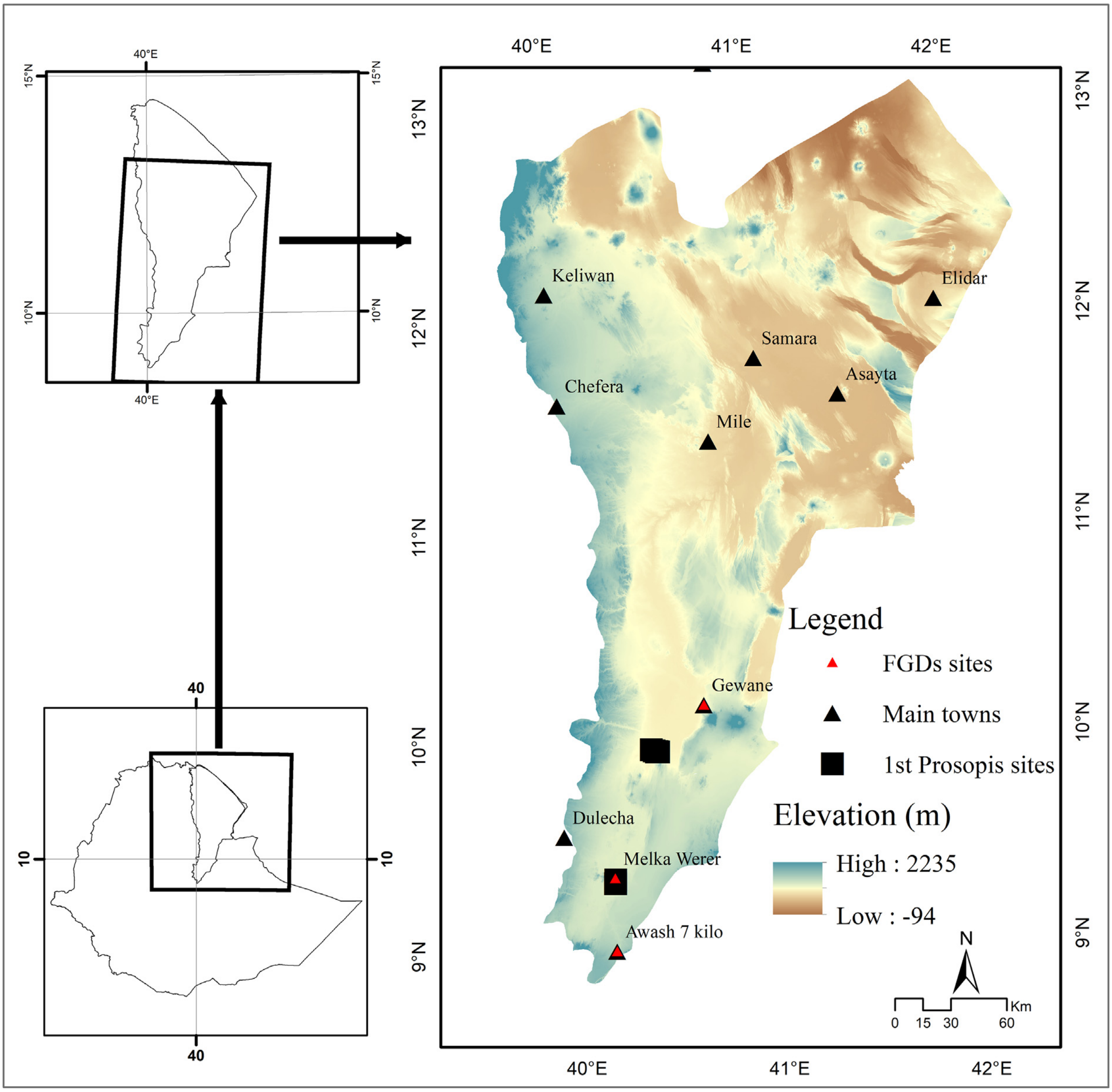


Fig. 1. Location of study area, background showing elevation ranging from - 94 m b.s.l. to 2235 m a.s.l.

temperature of 31 °C. The mean maximum temperature reaches up to 41 °C in June, and the mean minimum temperature ranges from 21 to 22 °C between November and December (Shiferaw et al., 2019a). The study area is located in the Awash Basin. The biome can be described as semi-arid dryland. Its vegetation cover consists of patches of scattered dry shrubs, acacia woodland (comprising different *Vachellia* species), bushland, grassland and wooded grassland (Engda, 2009). The study area has different soil types (from sand to heavy clays and stony soils), rocky outcrops, and a wide range of altitudes (ranging from 94 m below sea level to 2235 m a.s.l.).

Pastoralism is the dominant source of livelihood for the Afar people, but agro-pastoralism is being promoted by the government. In addition, governmental and private investments have promoted large-scale agricultural production in the Middle Awash Valley, mainly for cotton and sugar cane production. While the large areas of rain-fed grassland and shrubland in the region provide fodder for livestock, firewood and various natural products, including medicinal plants, the Afar people strongly depend on the floodplains of Awash River for grazing, particularly during the dry seasons, as well as for small-scale agriculture (Argaw, 2015). Parts of the arable land have been abandoned as a result of soil salinity, which has resulted from a combination of inappropriate irrigation practices and high evapotranspiration, and as a result of *Prosopis* invasion (Argaw, 2015).

Prosopis was first introduced in the Afar region in the late 1970s and early 1980s mainly for the purpose of water and soil conservation (Ayanu et al., 2014; Kebede and Coppock, 2015). Additional plantations were made between the 1980s and 1990s as shade and wind protection trees in villages and the raw material was used for firewood and fencing and building material (Ayanu et al., 2014). First problems arose soon thereafter, when the tree started invading croplands, grasslands, riverbanks and roadsides. *Prosopis* has been shown to reduce biodiversity, grazing potential and water supply, thereby causing significant impacts on the provision of key ecosystem services for (agro-) pastoralists (Shackleton et al., 2014). As a result, *Prosopis* has become a source of conflict among pastoralist groups in Ethiopia due to the resulting dwindling grazing land (Kebede and Coppock, 2015; Shiferaw et al., 2019a).

2.2. Methods

2.2.1. Remote sensing data and prepocessing

Satellite data were preprocessed using the Google Earth Engine (GEE) cloud computing environment. We worked with the Landsat Ecosystem Disturbance Adaptive Processing System (LEDAPS) surface reflectance products (Lu et al., 2002) of Landsat 5 TM, Landsat 7 ETM+ and Landsat 8 OLI, provided by USGS. All products are already geometrically co-registered, ortho-rectified, and atmospherically corrected. Images were provided together with a cloud mask and a quality assessment (QA) bands. We generated three different image collections having 30 m spatial resolution and consisting of the blue, green, red, near-infrared, as well as the two shortwave-infrared bands. The collections consist of pixels captured in the months of January and February, the dry season in Afar. In order to obtain cloud-free dry season composites of surface reflectance that captured the study areas' phenology consistently, we had to include two to three years of imagery to represent each point in time. Thus, for 1986, we selected imagery acquired between 1985 and 1987 giving preference to pixels captured in 1986; for 2000, we used data acquired between 1999 and 2001; and for 2017 we used data acquired from January/February 2016, 2017 and 2018, with the majority of pixels taken from 2017 imagery. Then, we used the LEDAPS QA band to remove clouded pixels, resulting in a stack of cloud-free pixel values for each pixel location, which we then reduced to a dry season composite by choosing the median pixel value for each pixel and spectral band. The use of the median pixel value ensured that outlier values (e.g., due to cloud shadows or clouds that were not previously removed by the QA band) were excluded. This was done for each optical Landsat band.

The reasons for our choice of season and years were (1) the dry season is being the optimum time to spectrally differentiate the evergreen *Prosopis* from other leaf-shedding indigenous vegetation, (2) the availability of cloud-free, good quality satellite images, and (3) capturing a) the early years of *Prosopis* presence (in 1986 *Prosopis* was just at plantation sites but started to invade areas outside the original plantations as it was introduced in the late 1970s and early 1980s (Kebede and Coppock, 2015), b) an intermediate point in time (2000), and c) the recent situation found in 2017.

2.2.2. Field data collection

For the classification of each dry season composite, reference data were collected for training and validation of each LULC type in the study area (Table 1). Reference data for 1986 were collected from aerial photographs captured in 1986. Careful attention was paid to collect only reference data using pure pixels of 30 × 30 m for each LULC type. Reference samples for 2000 were derived from the LULC maps of the Woody Biomass Project (Woody Biomass, 2000), and the Ethiopian Mapping Agency (EMA, 2003). Reference data for 2017 were collected directly from the field between September 2016 to January/February 2017 and 2018 using a handheld GPS (Garmin GPSMAP 60CSx). A total of 1847, 1998 and 2131 samples were collected for 1986, 2000 and 2017, respectively.

2.3. LULC classification, validation and dynamics

Field samples were partitioned and used for calibration (70%) as well as validation (30%) (Olecka, 2003; Oleksy, 2017). The LULC classifications for the three points in time were carried out using the field reference data set for calibration of a Random Forest (RF) classifier in R open source software (R Core Team, 2017). RF is one of the most known and versatile machine learning methods capable of performing both classification and regression tasks (CART: Breiman, 2001; Shiferaw et al., 2019b). It is widely used for environmental mapping and modeling applications (Cutler et al., 2007; Rodriguez-Galiano et al., 2012). Furthermore, 10-fold cross-validation was applied to assess model performance (Fushiki, 2009). Accuracy measures such as balanced accuracies, Kappa coefficients, sensitivity and specificity measures, user's and producer's accuracies were calculated using testing data set and error matrix was generated (Cleugh et al., 2012; Congalton, 1991).

Table 1
Land use/land cover types identified for analysis in the Afar National Regional State.

LULC type	Descriptions
Bareland	Areas without any vegetation either due to erosion or mismanagement (especially overgrazing); land cover permanently sand or rocks including volcanic black rocks or roads.
Cropland	Areas of land prepared for crop production. This category includes areas currently covered by crops, areas prepared for cultivation and fallow plots.
Grassland	Areas covered with natural grass and small shrubs, or dominated by grass, it includes areas used for communal grazing as well as a bare land that is seasonally grass-covered.
Natural forest	Mainly dominated by native *Vachellia* spp. >5 m of height and found in riverside/riverine forest ecosystems.
Prosopis	Areas invaded by *Prosopis* at different cover gradients (abundance), monoculture or mixed stands together with other vegetation but dominated by *Prosopis*.
Salt flats	Areas mostly in and around shallow lakes & ponds, used for salt production.
Settlements	Urban, build-up areas, temporary or permanent settlements.
Bush, shrub and woodland	Different types of pure or mixed trees such as palm trees, *Vachellia* woodland with grass as undergrowth. Canopy coverage is >10%. Shrublands and bush tickets consisting of mixed native shrubs of <3–5 m height and wooded bush thickets consisting of native species.
Waterbodies	Permanent lakes, and freshwater (rivers and streams). It also includes wetlands which dry up during the dry season, intermittent ponds and water points, perennial marshy areas, and man-made dams for hydroelectric and irrigation purposes.

After classification, LULC changes were calculated for three different time periods, i.e. 1986–2000, 2000–2017, and 1986–2017, using cross-tabulation (Eckert et al., 2017; Kindu et al., 2016; Zewdie and Csaplovics, 2015) and calculating percent changes for each LULC type over time (Gashaw et al., 2018; Kindu et al., 2016; Temesgen et al., 2018). Furthermore, class-specific gains, losses, and stable areas, as well as total change area and net changes of the total area analyzed were calculated (Alo and Pontius, 2008; Zewdie and Csaplovics, 2015). Finally, annual change rates were calculated for each LULC type following Puyravaud (2003) and Tilahun et al. (2014), i.e. the rate of change for a specific class was calculated by dividing the class-specific changes between two time intervals by the number of years between these two observed points in time.

2.4. Participatory assessment of perceived drivers of LULC change

To identify perceived main drivers of LULC changes over the last three decades, particularly those that might have influenced LULC changes prior to or during the years 1986, 2000 and 2017, focus group discussions (FGDs) were conducted with representatives of local communities from invaded and non-invaded areas in and around Awash Fentale, Amibara and Gewane districts. These three districts are located in the highly invaded areas of Zone 3, about 50 to 200 km away from each other (Fig. 1). In total, three different FGDs were conducted with six to ten persons each. The FGDs consisted of elderly, middle aged and young members of local communities, with a balanced composition in terms of gender. Most of the participants were pastoralists, agro-pastoralists, and agricultural extension officers (which are intermediaries between research and farmers). Each meeting lasted approximately 6 h. After an introduction to the purpose of the meeting, the group was invited to suggest possible drivers of LULC change. Those drivers perceived by the local community members as most influencing were then identified using the multi-criteria evaluation (MCE) technique in an analytic hierarchy process (AHP) method (Velasquez and Hester, 2013) applying pairwise comparison matrix (Saaty, 1994;

Saaty and Vargas, 2013). The MCE technique is used to explore how sensitively a driver varied while comparing it with another one on the Likert scale from 1 to 5 (1 = least important, 5 = most important). The next step was the testing of the sensitivity and consistency ratio in a pairwise comparison matrix (Saaty, 1994; Saaty and Vargas, 2013). Finally, weighting and sorting out the most important drivers were identified as the whole of the weighting criteria affects the outcome of the aggregation with respect to deviations in the preferences (Proctor and Drechsler, 2003). Hence, stable weighting was assigned for each potential driving factor so as to compare and identify those perceived as the most important ones (Saaty, 1994).

2.5. Ecosystem service values

The most common methods to calculate ESVs are based either on the simulated market approach (Chaudhary et al., 2015), the surrogate (proxy) market approach (Bai et al., 2013), or on the benefit transfer approach (Costanza et al., 1997; Niquisse et al., 2017). In this study, the benefit transfer approach was used to estimate values of ES of different LULC types and their changes (Costanza et al., 1997, 2014; Niquisse et al., 2017). The benefit transfer approach refers to the process of using existing values and other information from the original study site to estimate ESVs of other similar locations in the absence of site-specific valuation information (Bagstad et al., 2013; Niquisse et al., 2017).

We calculated the ESVs of the LULC types in ANRS taking two different approaches. First, we based our calculation of the ESV coefficients on regional estimates of ESVs using data provided by Kindu et al. (2016), who conducted a study on LULC and ESVs in Ethiopia using conservative estimates of ESV coefficients, which were based on values from studies conducted in areas similar to the geographical setting of our study area, which includes the main three ES: supply, regulation/monitoring and provision (Kindu et al., 2016) (Appendix 1a). To contrast this approach with one based on global estimates of ESV coefficients, we also calculated the ESVs of the LULC types in ANRS using the updated global coefficients provided by Costanza et al. (2014). Land use types such as bareland and settlement did not have a coefficient in some studies (Costanza et al., 1997; Kindu et al., 2016; Tolessa et al., 2017). Nevertheless, urban/settlement and salt flats were reported by Costanza et al. (2014) (Appendix 1a).

As no estimates of ESVs for *Prosopis* invaded areas were available, we estimated them based on information provided by a socio-economic study conducted in Afar region by Bekele et al. (2018a; unpubl. results) and interviews with key informants, which we define here as local people who have profound information about the study area and who already lived in the area some 30 years ago. For the regional assessment of the ESVs, we estimated the ESV coefficient for *Prosopis*-invaded areas by considering the estimated economic benefits of *Prosopis* for local stakeholders (Appendix 1b). For the approach based on global estimates of ESV coefficients, the ESV coefficient for *Prosopis*–invaded land was estimated from the average of two estimates of raw (wood) materials, i.e. that for tropical forests provided by Costanza et al. (2014) and that for woodland/shrubland provided by Temesgen et al. (2018) (Appendix 1b). The ESVs for all LULC types were calculated for each period using the following Equation (Costanza et al., 1997, 2014):

$$ESV = \sum[(Ak(VCk)]$$

where ESV = estimated ecosystem service value, Ak = the area (ha) of LULC type k, and VCk = the value coefficient (US$ ha^{-1} yr^{-1}) for LULC type k.

3. Results

3.1. LULC types, dynamics and rate of changes

With kappa accuracies between 88% (1986) and 92% (2017), RF classifier produced overall good to very good accuracies for the three classification points in time and the defined LULC types (Appendix 2; Viera and Garrett, 2005). Class-specific user's accuracies were at least 80%, with the exceptions of the class settlement in 1986 (79%) and 2000 (76%), the class waterbodies in 2000 (79%), the class *Prosopis* in 2000 (72%) and 2017 (79%). Producer's accuracies achieved also at least 80% for all LULC types, except for the class *Prosopis* in 2000 (75%).

The LULC change analysis for the period 1986–2017 revealed that cropland, *Prosopis*, salt flats, settlements and waterbodies have increased while bareland, grassland, natural forest and bush-shrub-woodland have decreased (Table 2; Fig. 2). While in 1986, the study area was dominated by bush-shrub-woodland, grassland and bareland, which made up about 99% of the study area, the share of these three classes decreased in 2017 to about 81%, with 14.5% of the study area newly covered by *Prosopis*. In absolute terms, *Prosopis* has invaded an area of about 965,407 ha over the last 31 years. It is the LULC type with the highest amount of change in the study area and reflects a mean annual invasion rate of 31,127 ha/year.

Our results suggest that the invasion did not increase linearly but was much faster after 2000 (1986–2000: 10,570 ha per year; and 2000–2017: 48,070 ha per year), which is also reflected when looking at the changes in percent area (Table 3). Considering changes in percent area within the last 31 years, we found that bareland and bush-shrub-woodland decreased by 5%, grassland by 9% (598,672 ha), and natural forest by 0.01% (994 ha), while *Prosopis* increased by 14.5%, which is about a 4000 times larger area than in 1986. Further, cropland, salt flats, settlements and waterbodies experienced positive net changes of 0.28%, 1.64%, 1.66%, and 0.58%, respectively (Table 3).

Between 1986 and 2000, *Prosopis* replaced mainly grassland and bush-shrub-woodland, while during the second time period (2000–2017) *Prosopis* also replaced bareland (Fig. 3). Overall, *Prosopis* gained some 208, 000 ha from bareland, 380,000 ha from bush-shrub-woodland, and 340,000 ha from grassland (Appendix 3).

3.2. Participatory assessment of perceived drivers of LULC change

The ten potential drivers of LULC change mentioned during the FGDs included indirect drivers, such as increasing human population and weak traditional law, as well as direct drivers, such as agricultural expansion, climate change and invasive species. The four potential drivers of LULC changes that were identified as the most influencing ones in the study area (importance >10%) were (1) climate change, (2) frequent droughts, (3) invasive species, and (4) weak traditional law (Fig. 4; for statistics on all driving factors mentioned by the interviewees, see Appendix 4). When asked why the local community members perceive 'climate change' and 'frequent droughts' as separate drivers of LULC change, they argued that climate change not only consisted of more frequent droughts but also other natural incidences such as change in temperature, heavy rainfall and flooding events. Regarding the perceived driver "weak traditional low", community members noted that in previous days local communities respected the customary law set by their forefathers and experienced throughout their life. However, with the reduction of resources due to land degradation and invasive species, the communal system of the pastoral areas allows use of their areas by people who come from anywhere within the region, rendering their communal land open-access.

3.3. Ecosystem service values

Based on regional estimates of ESV coefficients, the total ESVs in the study area dropped from 3110 million in 1986 to 2998 million in 2017. Based on global estimates of ESV coefficients, the total ESVs decreased from US$ 12,008 million to 10,916 million (Table 4). In both assessments, two natural LULC types lost large amounts of ESVs, namely grassland (US$ 175 to 2494 million) and bush-shrub-woodland (approx. US$ 323 million). On the other hand, *Prosopis*-invaded land (US$ 71 to 129 million) and waterbodies (US$ 311 to 481 million) increased in ESVs.

Table 2
Land use/land cover proportions for each type in hectares and percent share of the total area for 1986, 2000, 2017, and annual change rate during the study period.

LULC type	1986		2000		2017		Annual change rate
	ha	% share	ha	% share	ha	% share	(ha/year)
Bareland	2,023,898	30.7	1,882,955	28.6	1,694,915	25.4	−10,612
Bush-Shrub-Woodland	2,361,824	35.8	2,148,352	32.6	2,034,996	30.5	−10,543
Cropland	42,143	0.7	240,085	3.6	60,333	0.9	587
Grassland	2,195,201	32.1	2,060,777	31.3	1,596,529	23.9	−19,312
Natural forest	14,850	0.3	13,748	0.2	13,856	0.2	−32
Prosopis	241	0.01	148,218	2.3	965,166	14.5	31,127
Salt flats	13,830	0.2	17,053	0.3	138,174	2.1	4011
Settlement	1935	0.1	30,500	0.5	111,280	1.7	3527
Waterbodies	13,725	0.2	45,291	0.7	52,151	0.8	1240
Total	6,667,647	100	6,667,647	100.00	6,667,647	100	

The major losses of ESVs due to *Prosopis* invasion were from a change of bush-shrub-woodland and grasslands to *Prosopis*-dominated land. The losses of ESVs due to a change from grassland to *Prosopis* (340,000 ha; Fig. 3; Appendix 3) were estimated at US$ 74 to 1371 million based on regional and global estimates of a change in the ESV coefficient (ESV coefficient for grassland minus ESV coefficient for *Prosopis*), and the losses of ESVs due to change from bush-shrub-woodland to *Prosopis* (379,000 ha; Fig. 3) at US$ 323 to 346 million.

In ANRS, losses in ESVs primarily occurred in the areas along the major rivers, including the floodplains, and road corridors, where *Prosopis* is particularly invasive (Fig. 5). In contrast, areas closer to the highland escarpments, e.g. towards the Amhara National Regional State in the west, show relative stability in both LULC and ESVs. Areas which changed from bareland to waterbodies gained significantly in ESVs.

4. Discussions

Our assessment of LULC changes in the ANRS revealed a significant degradation of ES over the last 31 years and a high associated loss of ESVs. The most important change in land cover has occurred due to *Prosopis* invasion. *Prosopis* appears to have either directly replaced grassland and bush-shrub-woodland or has invaded bareland. Bareland may have historically exhibited low levels of vegetation or may have been degraded already before *Prosopis* started to invade. Hence, our results provide evidence that IAPS can be a key driver of LULC change and associated losses of ESVs at the regional scale.

4.1. LULC types, dynamics and rate of changes

The LULC change analysis revealed that LULC types particularly important for the ecosystem as well as peoples' livelihoods in the ANRS, namely grasslands, bush-shrub-woodland and to a lesser extent natural forests, have substantially decreased in the last 31 years. This reflects a general trend found in studies conducted in similar biomes in different parts of the world, e.g. in Australia (Cleugh et al., 2012), China (Li et al., 2007), Mozambique (Niquisse et al., 2017), as well as in different parts of Ethiopia (Gashaw et al., 2018; Hurni et al., 2005; Kindu et al., 2016;

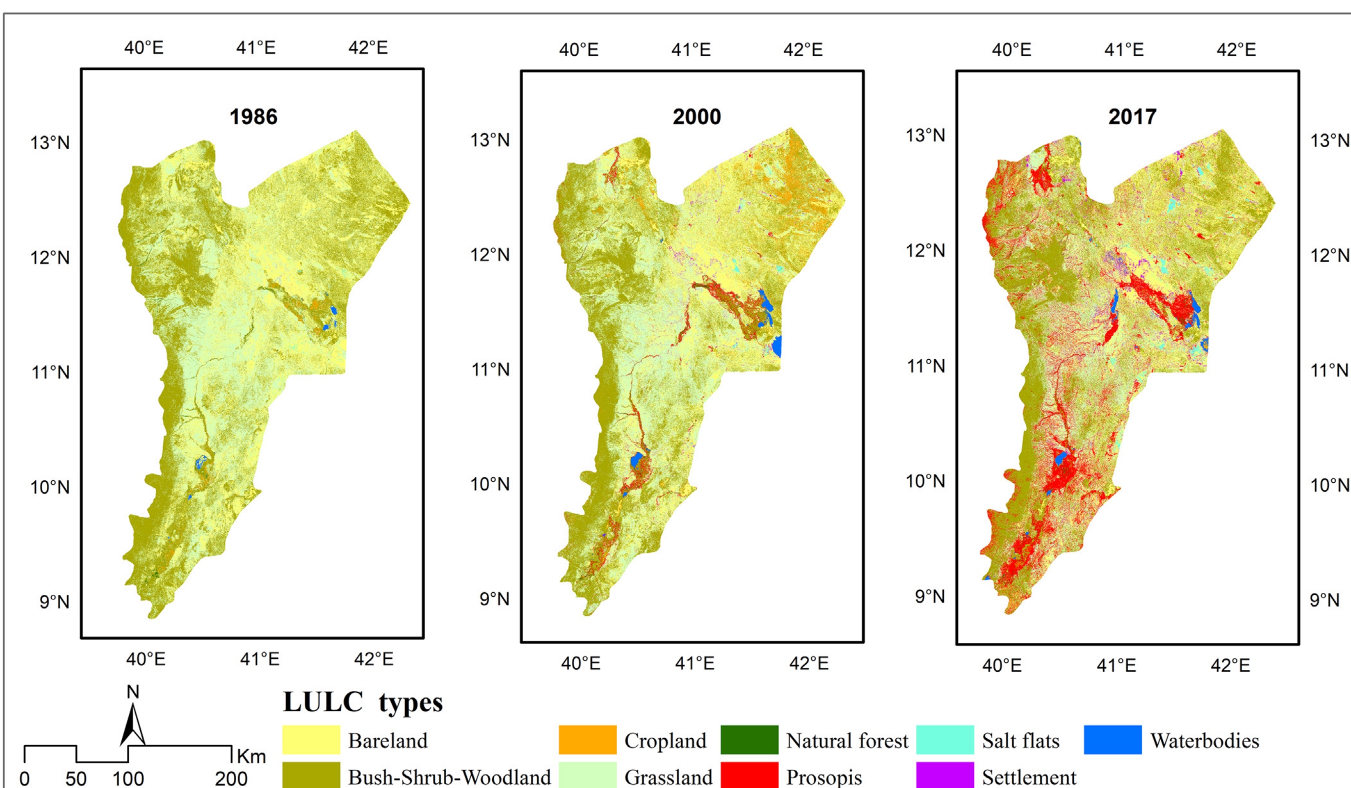


Fig. 2. Land use/cover classification results for 1986, 2000 and 2017 in the central and southern part of Afar National Regional State.

Table 3
The changes (1986–2017) of LULC types in hectares and percent shares.

LULC type	1986–2000		2000–2017		1986–2017	
	ha	%	ha	%	ha	%
Bareland	−140,943	−2.1	−188,040	−2.9	−328,982	−4.9
Bush-Shrub-Woodland	−213,471	−3.2	−113,356	−1.7	−326,827	−5.0
Cropland	197,943	3.0	−179,752	−2.7	18,192	0.3
Grassland	−56,824	−0.9	−464,248	−7.1	−598,672	−8.9
Natural forest	−1101	−0.02	−2953	−0.05	−994	−0.01
Prosopis	147,978	2.2	817,189	12.3	965,166	14.5
Salt flat	3224	0.05	43,519	0.7	124,344	1.6
Settlement	28,565	0.4	80,780	1.2	109,346	1.7
Waterbodies	31,567	0.5	6860	0.1	38,427	0.6

Tolessa et al., 2017; Tsegaye et al., 2010), but also at the global level (Costanza et al., 1997, 2014).

The Awash Basin is one of the river basins in Eastern Africa which supports significant numbers of pastoralists (Abule et al., 2005). However almost 600,000 ha (or 25%) of the grassland areas have been lost over the last 31 years (Fig. 3), largely due to invasion by *Prosopis* or a change to bareland. While indigenous species-dominated rangelands in Eastern Africa can occupy multiple stable states depending on fire frequency, rainfall, or grazing pressure (Anderies et al., 2002), invasion by *Prosopis* in grassland areas in the ANRS is hardly reversible, because established *Prosopis* trees are resistant to fire and can access groundwater (Dzikiti et al., 2013) also in areas with low or very low rainfall (Pasiecznik et al., 2001). Our results also suggest that grasslands were directly changed to *Prosopis*, without passing through an intermediate stage of degraded bareland. However, we acknowledge that the time interval between our LULC classifications was relatively long and that a short period of degradation, e.g. from perennial to annual grasslands, may have preceded *Prosopis* invasion in some cases. However, *Prosopis* has also started invading the Allideghi Wildlife Reserve, one of the last extensive grassland areas in ANRS, leading to a significant decrease in grass cover (Kebede and Coppock, 2015). Findings by Schachtschneider and February (2013) corroborate that *Prosopis* can also directly interfere with the survival of indigenous trees and shrubs.

The other major change in land cover is a shift from grassland to bareland and bush-shrub-woodland. As mentioned above, a shift between grass-dominated and shrub-dominated states is a natural process in African grasslands which is triggered, among others, by precipitation, fire and herbivory (Anderies et al., 2002). A shift of grassland to bareland, however, is likely to be a combination of overexploitation and climate change. Between 1960 and 2010, the population in Ethiopia has increased by 268% (Pricope et al., 2013), and this has translated into higher livestock stocking rates. In recent years, trends in

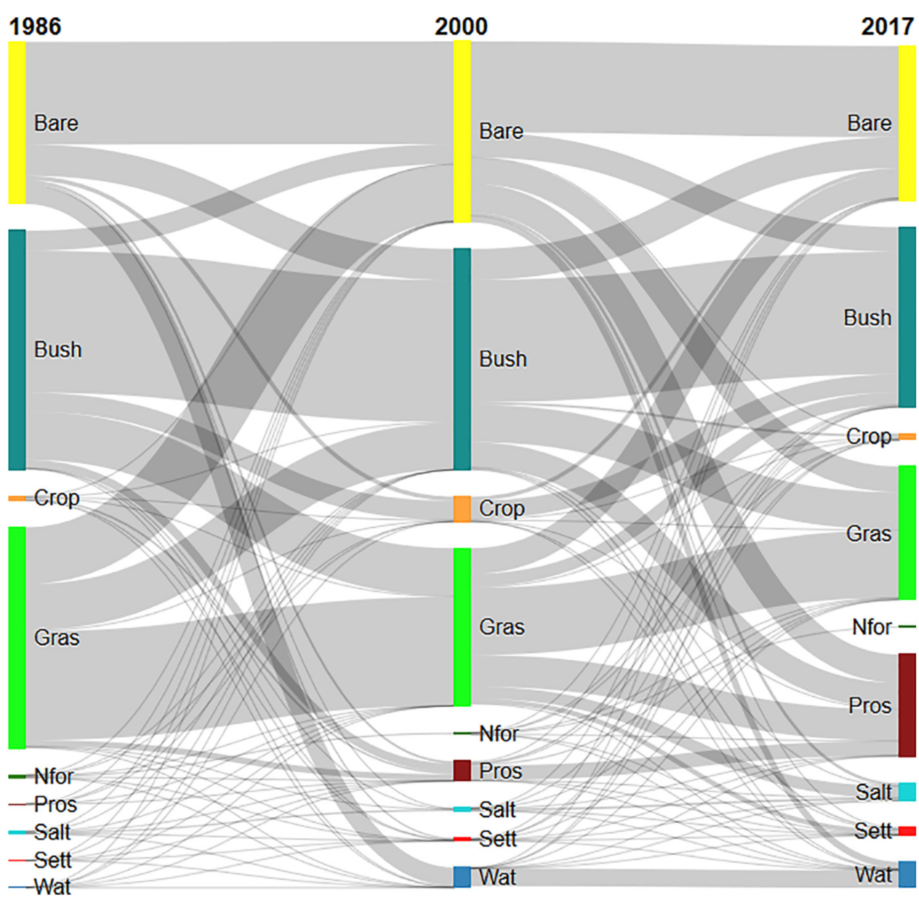


Fig. 3. Land use/cover types in ha: both stable and dynamic classes over the last 31 years between 1986–2000, 2000–2017, and 1986–2017. Bare = bareland, Bush = bush-shrub-woodland, Crop = cropland, Gras = grassland, Nfor = natural forest, Pros = *Prosopis*, Salt = salt flats, Sett = settlement, Wat = waterbodies.

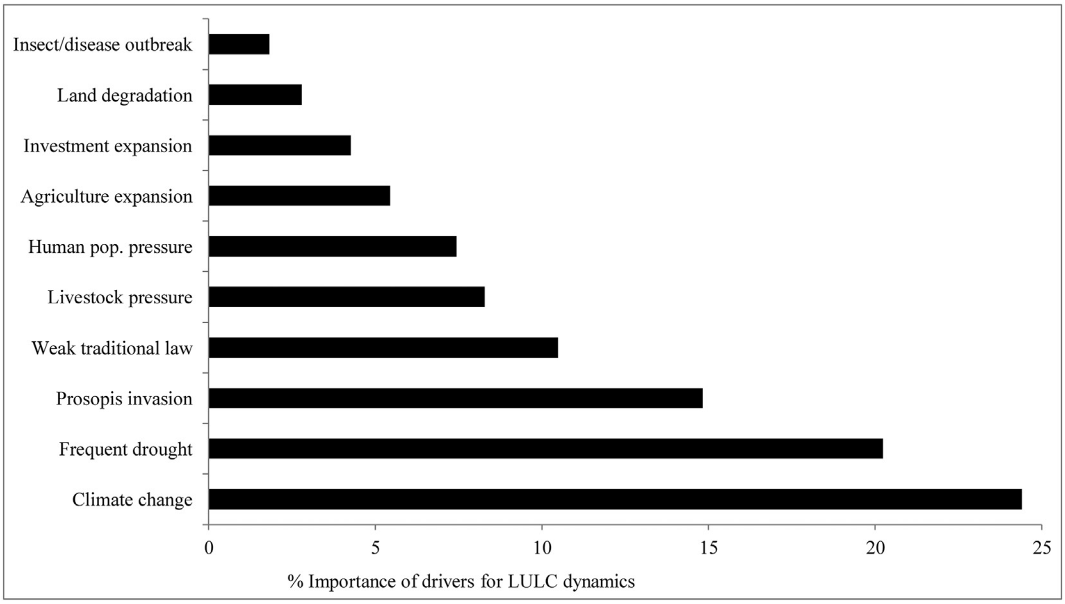


Fig. 4. Relative importance (%) of drivers of land use/cover changes in the Afar Region, Ethiopia.

livestock numbers have become more variable, but grazers have decreased and browsers increased (Yosef et al., 2013). Due to climate change, spring and summer rains in parts of Ethiopia have declined by 15–20% since the mid-1970s, and the observed warming across the entire country has further contributed to the increasing dryness (Funk et al., 2012). Hence, as perceived by the local communities, climate change is likely to negatively affect productivity of grasslands in the ANRS and, due to more frequent drought events, to reduce resilience of grasslands to grazing pressure. Yet, while grazing pressure was only ranked as a minor driver of LULC change by local stakeholders, the persistence of healthy grasslands in protected areas, such as Awash National Park, suggests that the main driver of change from grasslands to bareland over the last decades has been overgrazing by livestock (Abule et al., 2005). The increase of bareland is likely to have further facilitated *Prosopis* invasion, as Polley et al. (2003) showed that absence of competition with grass doubled emergence and almost tripled the survival of emergent seedlings of the congeneric species *Prosopis glandulosa* Torr.

The shift of some land classified as bareland in 2000 to grasslands in 2017 can probably be explained by the fact that the year 2000 was a drought year with low levels of rainfall (Haregeweyn et al., 2013; Viste et al., 2013). Hence, some grassland with very low vegetation cover, particularly those dominated by annual grasses, may have been misclassified as bareland in 2000.

Several 'anthropogenic' classes, such as cropland, settlements, salt flats and artificial waterbodies have also increased at the expense of natural vegetation cover. The increase in salt flats and waterbodies are associated with an increasing investment in the region in salt excavation and irrigated agriculture (Shiferaw, personal observation). As reported by Tsegaye et al. (2010), cropland area increased substantially during the 1990s and 2000s, but then decreased again thereafter. Similar trends of LULC changes towards more anthropogenic land use categories were found in other studies conducted in Eastern Africa, with cropland and settlements increasing at the expense of forests, shrubland, and grasslands (Eckert et al., 2017; Tolessa et al., 2017; Zewdie and Csaplovics, 2015). While the largest area under crop production (240,000 ha in the year 2000; Table 2) was far lower than the areas covered by grassland or bush-shrub-woodland, the impact of cropland on rural livelihoods in the ANRS is significant, because cropland is almost always located in the floodplains of Awash River and of its tributaries, i.e. in areas which were traditionally used by pastoralists as drought-season grazing areas. Crop production and the invasion of *Prosopis* are the main reasons why large areas of drought-season grazing land in the middle and lower Awash River Basin have been lost, which, in turn, has triggered ethnic conflicts in the downstream due to shortage of drought-season grazing land (Haji and Mohammed, 2013).

Our data indicate a dramatically increasing rate of spread of *Prosopis* in the ANRS, a phenomenon also reported from invasions of hybrid-

Table 4
Regional, global and average estimates of ecosystem service values (ESVs) in million US$ for each LULC type in 1986 and 2017; and the changes in ESVs in US$ between 1986 and 2017 in the Afar National Regional State, Ethiopia. Coefficients of ESVs were from Appendix 1a and * = ESV based on own calculations as explained in Appendix 1b.

LULC type	ESVs in 1986		ESVs in 2017		Changes in ESVs between 1986 & 2017		
	Regional estimate	Global estimates	Regional estimate	Global estimates	Regional estimate	Global estimates	Average estimates
Bareland	–	–	–	–	–	–	–
Bush-shrub-woodland	2331.1	2331.1	2008.5	2008.5	−322.6	−322.6	−322.6
Cropland	9.5	234.6	13.6	335.9	4.1	101.3	52.7
Grassland	643.2	9145.2	467.8	6651.1	−175.4	−2494.1	−1334.7
Natural forest	14.7	79.9	13.7	74.6	−1.0	−5.3	−3.2
*Prosopis**	0.02	0.03	71.4	129.4	71.4	129.3	100.4
Salt flats	–	32.3	–	323.2	–	290.8	145.4
Settlement	–	12.9	–	741.2	–	728.4	364.2
Waterbodies	111.2	171.7	422.6	652.5	311.4	480.8	396.1
Total	3109.7	12,007.9	2997.7	10,916.4	−112.0	−1091.4	−601.7

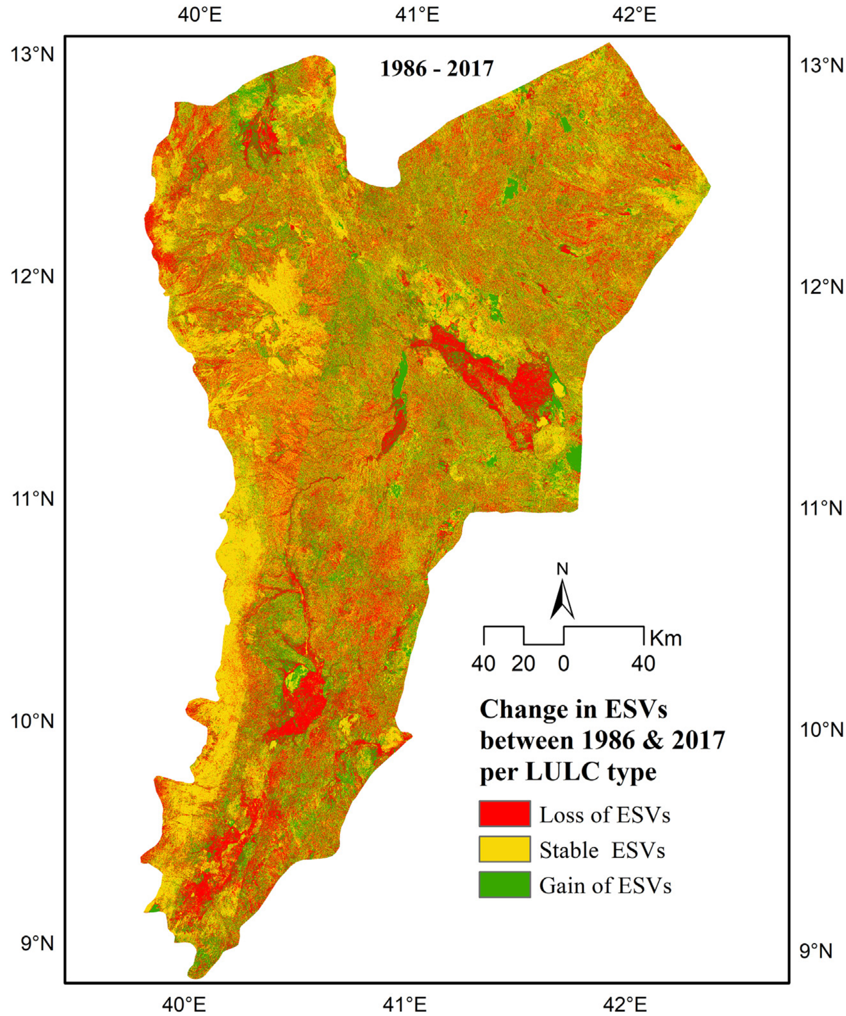


Fig. 5. Spatial distribution depicting loss, gain and stable ESVs in Afar National Regional State.

dominated *Prosopis* trees in South Africa (Shackleton et al., 2017; van Wilgen et al., 2012). This suggests that the rate of LULC changes in the ANRS will substantially increase over the next decades if *Prosopis* invasion is left uncontrolled and grassland not managed sustainably.

4.2. Participatory assessment of perceived drivers of LULC change

As perceptions of the environmental, economic and social factors driving environmental change play an important role in environmental decision-making (Meijer et al., 2015), it is important to understand the local stakeholders' perception of the factors affecting LULC change in Afar region, which in turn may inform to what extent they believe they can reduce the threats to the things they value. The most important drivers of LULC change perceived by the local community representatives in the FGDs were climate change, frequent droughts, invasive species, and weak traditional law. Climate change was also identified as the major cause for LULC changes, in other studies as it is a global phenomenon that causes but is also affected by LULC changes (Hansen et al., 2001; Pielke et al., 2002). Moreover, while climate change and invasive species are often treated as independent drivers, climate change is likely to exacerbate biological invasions (Pyke et al., 2008; Hulme, 2017). For example, climate change is likely to make it easier for some invasive species to establish and survive, and may accelerate the spread of successfully established invasive species by disrupting native communities and ecosystems (Eiswerth et al., 2005; Hulme, 2017) or by increasing atmospheric CO_2 concentrations, which may promote *Prosopis* recruitment in grasslands (Jackson et al., 2002; Polley et al., 2003). Hence, management of invasive species could also increase the resilience of ecosystems to climate change.

The second important driver of LULC change identified by FDGs was frequent droughts in the study area. Similarly, Tsegaye et al. (2010) identified drought as the first major driver of LULC change in northern part of the ANRS. While droughts have always occurred in the ANRS, they now occur at shorter intervals than in recent years (El Kenawy et al., 2016; Viste et al., 2013). Pastoralists from the ANRS mentioned that in former times grasslands and the number of livestock used to recover quickly after drought events, but that this has changed due to the combination of climate change and *Prosopis* invasion (Haji and Mohammed, 2013). Not surprisingly, the stakeholders are fully aware of the role of *Prosopis* in affecting LULC changes in the study area, and their emphasis on its impacts on bush-shrub-woodland and grassland during the FGDs is in line with our findings based on LULC changes in the ANRS (Shiferaw et al., unpubl. data).

Local communities claimed that traditional (customary) laws have become weak and that this has contributed to the observed LULC changes in the area. For example, rotational programs for seasonal grazing areas had been planned and maintained for the specified communities for extended periods. However, in recent times, communities but also people coming from other parts of Afar region or other regions use areas banned for grazing at any time, thereby undermining the efficacy of grassland's natural restoration capacities (Tessema et al., 2016).

In the northern parts of the ANRS, which are relatively free from *Prosopis* invasion, local communities ranked land degradation as the fourth most important factor to cause LULC change, besides drought, rainfall variability and firewood/timber (Tsegaye et al., 2010). In our study, however, which focused on the southern and the central parts of the ANRS, land degradation was ranked comparably low. This may be because the local stakeholders believe that land degradation by itself is not a primary cause for LULC change but a side effect of others, higher ranked direct drivers such as climate change, frequent droughts, and *Prosopis* invasion.

The other drivers of LULC change identified were an increase in large scale agricultural investments in the study area, namely in sugar plantations and a large-scale cotton farms also lead to LULC changes. Population pressure, overgrazing and expansion of salt excavation and agricultural irrigation projects were additional factors mentioned as contributors to LULC change, although to a lesser extent. The low ranking of population pressure as a potential driver of LULC change sounds somehow surprising, given that Ethiopia is one of the World's most populous countries and has a high population growth rate of 2.4% (CSA, 2016). However, local people in the ANRS may consider population pressure as a minor driver of LULC change because they weigh indirect drivers lower than direct drivers of LULC change.

4.3. Ecosystem service values

Our findings suggest that the study area, which extends over an area of 6.67 million ha, has lost a substantial amount of ESVs over the last 31 years, no matter which estimates of ESV coefficients (regional or global values) were used. As expected, the conservative regional approach, which only considered values from tropical areas of LULC types similar to the geographical setting of the study area (Ethiopian highlands: Kindu et al., 2016) generated considerably lower estimates of ESVs lost than that based on global average estimates of LULC types (Costanza et al., 2014). As the estimate of the ESVs for the *Prosopis* class is based on information from the original study site (Bekele et al., 2018a, 2018b; see Appendix 1b), the regional estimate can be considered as a more coherent one, since all ESV estimates are based on studies from the study area itself or areas from the same biome (Olson et al., 2001) but not necessarily from the same topographic settings. Kindu et al. (2016) estimated that in a 10,000 ha area in the Ethiopian highlands, ESVs had dropped over the last 40 years by US$ 19.3 million (or 48.3 US$ ha^{-1} y^{-1}) when using their own coefficients of ESVs. In our study, regional estimates of ESV coefficients resulted in a loss of US$ 112 million (or 17.35 US$ ha^{-1} y^{-1}). The lower estimates of ESV for ANRS compared to those reported by Kindu et al. (2016) for Ethiopian highlands are reasonable as highlands have better agro-climatic conditions for increased delivery of fresh water, food, fodder and other ecosystems services. Hence, the lower ESVs in ANRS could be explained by two reasons: a higher area of bareland with no ESVs as well as a lower overall productivity of the semi-arid areas in the region.

In the assessment based on regional estimates of ESV coefficients, the reduction of bush-shrub-woodland contributed most to the loss of ESVs, followed by the reduction of grasslands. In contrast, in the assessment based on global estimates of ESV coefficients the loss of grasslands contributed seven times more to the overall loss of ESVs in the ANRS than bush-shrub-woodland. This can be explained by the recent raise in awareness of the potential of healthy grasslands in providing not only provisioning but also significant amounts of supporting and regulating ES (Veldman et al., 2015a, 2015b). Kindu et al.'s (2016) conservative estimate of the ESV coefficient for grassland was derived from the first global study on ESV coefficients published by Costanza et al. (1997). From the first to the second worldwide assessment of the values of ES (Costanza et al., 1997, 2014), the average ESVs per ha for grassland/rangeland increased 13-fold, while forests only increased 2-fold. With regard to this recent increase in the valuation of grasslands, the assessment based on global estimates of ESV coefficients may therefore reflect the real loss of ESVs in the ANRS more accurately.

An additional factor that may further increase the negative effects of *Prosopis* invasion on the overall ESVs of the invaded range is its impact on water availability and accessibility. *Prosopis* is a deep-rooted tree, which draws water from the soil and deep underwater (Dzikiti et al., 2013), thereby affecting availability of water for permanent bunchgrasses and other keystone species of healthy, old-growth grasslands (Veldman et al., 2015a, 2015b). This is particularly problematic in the ANRS where the evergreen *Prosopis* consumes water in a water-limited ecosystem throughout the year. On the other hand, *Prosopis* creates physical barriers by invading riversides with its dense and thicket shrubs, thereby preventing accessibility of the rivers to domestic and wild animals, and increases densities of vectors of human diseases (Muller et al., 2017). These EDS due to *Prosopis* invasion provide additional evidence that the results obtained in our study can be considered as an example where a single invasive alien plant species can seriously affect the ESVs of a whole region. In an attempt to estimate the benefits and costs of invasive *Prosopis* species and hybrids in the Northern Cape, South Africa, Wise et al. (2012) concluded that it is beneficial to control *Prosopis* in floodplain habitats, similar to those along Awash river in ANRS, largely because it avoids groundwater losses. They estimated that the benefits of controlling *Prosopis* in the floodplains amount, over a 30 year period, to US$ 56–137 million in a slow spread-rate and to US$ 122–376 million in a rapid spread-rate scenario (Wise et al., 2012).

Moreover, when an agricultural land is invaded by *Prosopis*, it affects the livelihoods of the user communities in terms of increasing clearing cost and time for land preparation, which increase cost of production and inflated market price for agricultural produces in the study area and outside. Similarly, where rangelands/grasslands are replaced by *Prosopis*, the livelihoods of pastoralists are affected in multiple ways: (1) invasion affects availability of indigenous forage and pasture by reducing available native forage as well as water resources (Shackleton et al., 2015b), (2) livestock are forced to travel long distances in search for food and water, and only those animals that are able to travel long distances can survive especially during drought season as it happened in the lowlands of the country in 2015; and travelling long distance out of their territories creates conflict among user groups of different tribes, (3) livestock number and diversity are affected as grasslands are diminished or are changed to *Prosopis* invaded areas, and commonly, grazers are replaced by browsers as well as nutrition values and quality of pastures of diversified feed types are replaced into mono-crop of *Prosopis* pods, which in turn affect the health and productivity of livestock in terms of both meat and milk productions (Shiferaw, personal communication with key informants); and (4) wild animals hide themselves in the invaded areas and attack domestic animals.

It is likely that ESVs from cropland will also further increase in the future due to a growing need for food and thus expansion of crop production in order to nourish the increasing population (Niquisse et al., 2017). Several large scale agricultural investments have already been established in recent years in the ANRS, mainly along the major water courses. This development will have a substantial impact on future LULC changes and ESVs. In addition, the expansion of investment programs in the northern part of the study area on salt flats is expected to consume large areas of seasonal grasslands, bareland and waterbodies (Shiferaw, personal observation).

The average annual loss in ESVs in our study area is more than fourfold of the annual budget plan of the whole ANRS in 2016/17 (BoFED, 2017). This suggests that changes in ESVs should be considered as one of the indicators of stability of socio-ecological systems, and of human welfare, and hence their assessment should be considered as a policy instrument (Niquisse et al., 2017) in the ANRS as well as elsewhere exhibiting with the same challenges.

5. Conclusions

Our study provides evidence that invasive species can be a main driver of LULC change and associated losses of ESVs at the regional scale. As *Prosopis* was introduced in the ANRS in the late 1970s and early 1980s, the time period assessed in our study covers the whole invasion history of *Prosopis* in the area. Within this period, *Prosopis* has invaded approximately 1 million ha in the study site. In parallel, the ANRS has experienced a significant loss of grasslands, bush-shrub-woodland and riverine natural forest. While parts of the grasslands present in the 1980s have been mainly invaded by *Prosopis*, other drivers such as unsustainable management and climate change, either in isolation or in combination, are likely to have also contributed to the loss of grasslands, with serious consequences for the pastoralists and agro-pastoralists who directly depend on ES provided by grasslands (Bekele et al., 2018a, 2018b; Shackleton et al., 2015b).

Mainstreaming ES and its values into policy and decision making is dependent on the availability of spatially explicit information on the state and trends of ecosystems and their services (Maes et al., 2012). As *Prosopis* is likely to further spread in ANRS if left uncontrolled (Shiferaw et al., 2019a), our results strongly advocate for a rapid implementation of the National *Prosopis* Management Strategy designed for Ethiopia (MoLF, 2017). Moreover, there is a need for designing restoration and/or rehabilitation programs to make the area resilient to climate change, frequent drought as well as invasion species impacts so that sustainable ES and functions are maintained. Together with the implementation of sustainable grassland and bush-shrub-woodland management practices, halting or slowing down the *Prosopis* invasion will be fundamental for preserving or even restoring the remaining ESVs in the ANRS and probably other invaded regions in Eastern Africa.

Acknowledgements

We are grateful for financial support from the Swiss Programme for Research on Global Issues for Development (r4d), funded by the Swiss National Science Foundation (SNSF) and the Swiss Agency for Development and Cooperation (SDC), for the project "Woody invasive alien species in East Africa: Assessing and mitigating their negative impact on ES and rural livelihood" (Grant Number: 400440_152085). Urs Schaffner was supported by CABI with core financial support from its member countries (see http://www.cabi.org/about-cabi/who-we-work-with/key-donors/). We thank the staff of Addis Ababa University's Water and Land Resource Center for their support and facilitation services and we also thank Dr. Gizaw Desta and Dr. Mengistie Kindu for commenting on an earlier version of the manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2019.04.220.

References

Abule, E., Snyman, A., Smit, N., 2005. Comparisons of pastoralists perceptions about rangeland resource utilisation in the Middle Awash Valley of Ethiopia. J. Environ. Manag. 75 (1), 21–35.

Alo, A., Pontius, G., 2008. Identifying systematic land-cover transitions using remote sensing and GIS: the fate of forests inside and outside protected areas of Southwestern Ghana. Environ. Plan. B Plan. Des. 35, 280–295. https://doi.org/10.1068/b32091.

Anderies, M., Janssen, A., Walker, H., 2002. Grazing management, resilience, and the dynamics of a fire-driven rangeland system. Ecosystems 5 (1), 23–44.

Argaw, T., 2015. Impacts of utilizing invasive Prosopis juliflora (Swartz DC.) on rural household economy at Gewane District, Afar Regional. J. Econ. Sustain. Dev. 6, 81–98.

Arowolo, O., Deng, X., Olatunji, A., Obayelu, E., 2018. Assessing changes in the value of ecosystem services in response to land-use/land-cover dynamics in Nigeria. Sci. Total Environ. 636, 597–609.

Ayanu, Y., Jentsch, A., Müller-Mahn, D., Rettberg, S., Romankiewicz, C., Koellner, T., 2014. Ecosystem engineer unleashed: Prosopis juliflora threatening ecosystem services? Reg. Environ. Chang. 15, 155–167. https://doi.org/10.1007/s10113-014-0616-x.

Bagstad, J., Johnson, W., Voigt, B., Villa, F., 2013. Spatial dynamics of ecosystem service flows: a comprehensive approach to quantifying actual services. Ecosyst. Serv. 4, 117–125. https://doi.org/10.1016/j.ecoser.2012.07.012.

Bai, Z., Dent, D., Wu, Y., de Jong, R., 2013. Land degradation and ecosystem services. In: Lal, R., Lorenz, K., Hüttl, R.F., Schneider, B.U., von Braun, J. (Eds.), Ecosystem Services and Carbon Sequestration in the Biosphere. Springer Netherlands, Dordrecht, pp. 357–381.

Bekele, K., Haji, J., Legesse, B., Shiferaw, H., Schaffner, U., 2018a. Impacts of woody invasive alien plant species on rural livelihood: generalized propensity score evidence from Prosopis spp. invasion in Afar Region in Ethiopia. Pastoralism: Research, Policy and Practice 8 (28), 1–17. https://doi.org/10.1186/s13570-018-0124-6.

Bekele, K., Haji, J., Legesse, B., Schaffner, U., 2018b. Economic impacts of Prosopis spp. invasions on dryland ecosystem services in Ethiopia and Kenya: evidence from choice experimental data. J. Arid Env. 158, 9–18.

BoFED, 2017. Annual Planning for 2015/16. Bureau of Finance and Economic Development. MoFED, Addis Ababa.

Braat, C., Groot, D., 2012. The ecosystem services agenda: bridging the worlds of natural science and economics, conservation and development, and public and private policy. Ecosyst. Serv. 1, 4–15. https://doi.org/10.1016/j.ecoser.2012.07.011.

Breiman, L., 2001. Random forest. Mach. Learn. 1, 1–33. https://doi.org/10.1017/CBO9781107415324.004.

Chaudhary, S., McGregor, A., Houston, D., Chettri, N., 2015. The evolution of ecosystem services: a time series and discourse-centered analysis. Environ. Sci. Pol. 54, 25–34. https://doi.org/10.1016/j.envsci.2015.04.025.

Cleugh, A., Leuning, R., Mu, Q., Running, W., Teixeira, C., Bastiaanssen, et al., 2012. Developing land cover and land use data sets for the Australian continent – a collaborative approach. Remote Sens. Environ. 1, 343–354. https://doi.org/10.1016/j.proenv.2012.01.162.

Congalton, G., 1991. A review of assessing the accuracy of classifications of remotely sensed data. Remote Sens. Environ. 37, 35–46. https://doi.org/10.1016/0034-4257(91)90048-B.

Costanza, R., Arge, R., Groot, R. De, Farberk, S., Grasso, M., Hannon, B., et al., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260. https://doi.org/10.1038/387253a0.

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., et al., 2014. Changes in the global value of ecosystem services. Glob. Environ. Chang. 26, 152–158. https://doi.org/10.1016/j.gloenvcha.2014.04.002.

Crespin, J., Simonetti, A., 2016. Loss of ecosystem services and the decapitalization of nature in El Salvador. Ecosyst. Serv. 17, 5–13. https://doi.org/10.1016/j.ecoser.2015.10.020.

CSA, 2016. Annual Demographic Survey Report. Central Statistical Agency (CSA), Addis Ababa.

Cutler, R., Edwards, C., Beard, H., Cutler, A., Hess, T., Gibson, J., Lawler, J., 2007. Random forests for classification in ecology. Ecology 88, 2783–2792. https://doi.org/10.1890/07-0539.1.

Dzikiti, S., Schachtschneider, K., Naiken, V., Gush, M., Moses, G., Le Maitre, C., 2013. Water relations and the effects of clearing invasive Prosopis trees on groundwater in an arid environment in the Northern Cape, South Africa. J. Arid Environ. 90, 103–113. https://doi.org/10.1016/j.jaridenv.2012.10.015.

Eckert, S., Kiteme, B., Njuguna, E., Zaehringer, G., 2017. Agricultural expansion and intensification in the foothills of Mount Kenya: a landscape perspective. Remote Sens. 9, 1–20. https://doi.org/10.3390/rs9080784.

Eiswerth, E., Darden, D., Johnson, S., Agapoff, J., Harris, R., 2005. Input–output modeling, outdoor recreation, and the economic impacts of weeds. Weed Sci. 53 (1), 130–137.

El Kenawy, M., McCabe, F., Vicente-Serrano, M., López-Moreno, I., Robaa, M., 2016. Cambios en la frecuencia y severidad en las sequías hidrológicas de Etiopía entre 1960–2013. Cuad. Investig. Geogr. 42, 145–166. https://doi.org/10.18172/cig.2931.

EMA, 2003. Ethiopian Mapping Agency (EMA). Land Use and Land Cover Mapping Using Landsat ETM+ (2000–2001). Ababa, Addis.

Engda, G., 2009. Spatial and Temporal Analysis of Prosopis Juliflora (Swarz) DC Invasion in Amibara Woreda of the Afar NRS Spatial and Temporal Analysis of Prosopis Juliflora (Swarz) DC Invasion in Amibara Woreda of the Afar NRS. MSc Thesis. AAU.

Funk, C., Rowland, J., Eilerts, G., Kebebe, E., Nigist, B., White, L., Gideon, G., 2012. A Climate Trend Analysis of Ethiopia. USAID/FEWSNET https://www.usaid.gov/sites/default/files/documents/1860/A%20Climate%20Trend%20Analysis%20of%20Ethiopia.pdf.

Fushiki, T., 2009. Estimation of prediction error by using K-fold cross-validation. T. Stat Comput. 21, 137–146. https://doi.org/10.1007/s11222-009-9153-8.

Gashaw, T., Tulu, T., Argaw, M., Worqlul, A.W., Tolessa, T., Kindu, M., 2018. Estimating the impacts of land use/land cover changes on ecosystem service values: the case of the Andassa watershed in the Upper Blue Nile basin of Ethiopia. Ecosyst. Serv. 31, 219–228. https://doi.org/10.1016/j.ecoser.2018.05.001.

Goldman-Benner, L., Benitez, S., Boucher, T., Calvache, A., Daily, G., Kareiva, P., et al., 2012. Water funds and payments for ecosystem services: practice learns from theory and theory can learn from practice. Oryx 46, 55–63. https://doi.org/10.1017/S0030605311001050.

Grêt-Regamey, A., Weibel, B., Kienast, F., Rabe, S., Zulian, G., 2015. A tiered approach for mapping ecosystem services. Ecosyst. Serv. 13, 16–27. https://doi.org/10.1016/j.ecoser.2014.10.008.

Haji, J., Mohammed, A., 2013. Economic impact of Prosopis juliflora on agropastoral households of Dire Dawa Administration. Ethiopia. African J. Agric. 8, 768–779. https://doi.org/10.5897/AJAR12.014.

Hansen, A., Neilson, R., Dale, V., Flather, C., Iverson, L., Currie, D., et al., 2001. Global change in forests: responses of species, communities, and biomes: interactions between climate change and land use are projected to cause large shifts in biodiversity. BioScience 51 (9), 765–779. https://doi.org/10.1641/0006-3568(2001)051[0765:GCIFRO]2.0.CO;2.

Haregeweyn, N., Tsunekawa, A., Tsubo, M., Meshesha, D., Melkie, A., 2013. Analysis of the invasion rate, impacts and control measures of *Prosopis juliflora*: a case study of Amibara District, Eastern Ethiopia. Environ. Monit. Assess. 185, 7527–7542. https://doi.org/10.1007/s10661-013-3117-3.

Hulme, E., 2017. Climate change and biological invasions: evidence, expectations, and response options. Biol. Rev. 92 (3), 1297–1313.

Hulme, E., Burslem, P., Dawson, W., Edward, E., Richard, J., Trevelyan, R., 2013. Plant Invasions in Protected Areas. Springer, Dordrecht Heidelberg New York London https://doi.org/10.1007/978-94-007-7750-7 (ISBN 978-94-007-7749-1 ISBN 978-94-007-7750-7 (eBook)).

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. https://doi.org/10.1659/0276-4741(2005)025[0147:TIOCIP]2.0.CO;2.
Jackson, R.B., Banner, J.L., Jobbágy, E.G., Pockman, W.T., Wall, D.H., 2002. Ecosystem carbon loss with woody plant invasion of grasslands. Nature 418 (6898), 623–626.
Kebede, T., Coppock, L., 2015. Livestock-mediated dispersal of Prosopis juliflora imperils grasslands and the endangered Grevy's zebra in Northeastern Ethiopia. Rangel. Ecol. Manag. 68, 402–407. https://doi.org/10.1016/j.rama.2015.07.002.
Keller, P., Lodge, M., Lewis, M., Shogren, J., 2010. Bioeconomics of Invasive Species: Integrating Ecology, Economics, Policy, and Management. Oxford Univer.
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. https://doi.org/10.1016/j.scitotenv.2015.12.127.
Kremer, P., Hamstead, Z., Haase, D., McPhearson, T., Frantzeskaki, N., Andersson, E., et al., 2016. Key insights for the future of urban ecosystem services research. Ecol. Soc. 21 (29). https://doi.org/10.5751/ES-08445-210229.
Le Maitre, C., Kotzee, I.M., O'Farrell, J., 2014. Impacts of land-cover change on the water flow regulation ecosystem service: invasive alien plants, fire and their policy implications. Land Use Policy 36, 171–181.
Lefcheck, S., Byrnes, K., Isbell, F., Gamfeldt, L., Griffin, N., Eisenhauer, N., et al., 2015. Nature communications biodiversity enhances ecosystem multifunctionality across trophic levels and habitats. Nat. Commun. 6, 1–7. https://doi.org/10.1038/ncomms7936.
Leh, D., Matlock, D., Cummings, C., Nalley, L., 2013. Quantifying and mapping multiple ecosystem services change in West Africa. Agric. Ecosyst. Environ. 165, 6–18.
Li, Q., Dong, M., Cui, Y., Zhang, L., Cui, G., He, 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. https://doi.org/10.1007/s10661-006-9344-0.
Liu, C., White, M., Newell, G., 2009. Measuring the accuracy of species distribution models: a review. 18th World IMACS/MODSIM Congr. Cairns, Aust. 13–17 July 2009, pp. 4241–4247.
Lowe, S., Browne, M., Boudjelas, S., De Poorter, M., 2000. 100 of the world's worst invasive alien species: a selection from the global invasive species database. Invasive Species Specialist Group. vol. 12. IUCN, Auckland, New Zeeland, p. 12.
Lu, D., Mausel, P., Brondizio, E., Moran, E., 2002. International Journal of Assessment of atmospheric correction methods for Landsat TM data applicable to Amazon basin LBA. Int. J. Remote Sens. 23, 2651–2671. https://doi.org/10.1080/01431160110109642.
Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Grizzetti, B., et al., 2012. Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst. Serv. 1, 31–39. https://doi.org/10.1016/j.ecoser.2012.06.004.
Meijer, S.S., Catacutan, D., Ajayi, O.C., Sileshi, G.W., Nieuwenhuis, M., 2015. The role of knowledge, attitudes and perceptions in the uptake of agricultural and agroforestry innovations among smallholder farmers in sub-Saharan Africa. Int. J. Agric. Sustain. 13 (1), 40–54.
MoLF, 2017. National Strategy for *Prosopis juliflor*a Management. Ministry of Livestock and Fisheries of Ethiopia, Addis Ababa, Ethiopia.
Muller, G.C., Junnila, A., Traore, M.M., Traore, S.F., Doumbia, S., Sissoko, F., Dembele, S.M., Schlein, Y., Arheart, K.L., Revay, E.E., Kravchenko, V.D., Witt, A., Beier, J.C., 2017. The invasive shrub Prosopis juliflora enhances the malaria parasite transmission capacity of Anopheles mosquitoes: a habitat manipulation experiment. Malar. J. 16 (1), 237. https://doi.org/10.1186/s12936-017-1878-9.
Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, R., et al., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front. Ecol. Environ. 7, 4–11. https://doi.org/10.1890/080023.
Niquisse, S., Cabral, P., Rodrigues, Â., Augusto, G., 2017. Ecosystem services and biodiversity trends in Mozambique as a consequence of land cover change. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 13, 297–311. https://doi.org/10.1080/21513732.2017.1349836.
Olecka, A., 2003. Evaluating classifiers' performance in a constrained environment. Association for Computing Machinery, KDD 2002, 605–612. https://doi.org/10.1145/775107.775137.
Oleksy, T., 2017. Machine Learning Methods for Mood Disorder Decision Support. MSc Thesis. Bergen University College, Norway, p. 126.
Olson, M., Dinerstein, E., Wikramanayake, E., Burgess, D., Powell, N., Underwood, C., et al., 2001. Terrestrial ecoregions of the world: a new map of life on Earth. Bioscience 51 (11), 933–938 [online] URL:. http://worldwildlife.org/publications/terrestrial-ecoregions-of-the-world.
Paini, R., Sheppard, W., Cook, C., De Barro, J., Worner, P., Thomas, B., 2016. Global threat to agriculture from invasive species. Proc. Natl. Acad. Sci. 113 (27), 7575–7579.
Pasiecznik, M., Felker, P., Harris, C., Harsh, N., Cruz, G., Tewari, C., Cadoret, K., Maldonado, J., 2001. The Prosopis juliflora - Prosopis pallida Complex: A Monograph. HDRA, Coventry, UK, p. 172.
Pejchar, L., Mooney, A., 2009. Invasive species, ecosystem services and human well-being. Trends Ecol. Evol. 24, 497–504. https://doi.org/10.1016/j.tree.2009.03.016.
Pielke, R., Marland, G., Betts, R., Chase, T., Eastman, J., Niles, J., Niyogi, D., Running, S., 2002. 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. Lond. A. 360, 1705–1719. https://doi.org/10.1098/rsta.2002.1027.
Polley, W., Johnson, H., Tischler, C., 2003. Woody invasion of grasslands: evidence that CO_2 enrichment indirectly promotes establishment of *Prosopis glandulosa*. Plant Ecol. 164 (1), 85–94. https://doi.org/10.1023/A:1021271226866.
Pricope, N., Husak, G., Lopez-Carr, D., Funk, C., Michaelsen, J., 2013. The climate-population nexus in the East African Horn: emerging degradation trends in rangeland and pastoral livelihood zones. Glob. Environ. Chang. 23, 1525–1541.
Proctor, W., Drechsler, M., 2003. Deliberative multi-criteria evaluation: a case study of recreation and tourism options in Victoria. CSIRO Sustainable Ecosystems. Canberra, Australia.
Puyravaud, P., 2003. Standardizing the calculation of the annual rate of deforestation. For. Ecol. Manag. 177, 593–596. https://doi.org/10.1016/S0378-1127(02)00335-3.
Pyke, R., Thomas, R., Porter, D., Hellmann, J., Dukes, S., Lodge, M., et al., 2008. Current Practices and Future Opportunities for Policy on Climate Change and Invasive Species. 22, pp. 585–592. https://doi.org/10.1111/j.1523-1739.2008.00956.x.
R Core Team, 2017. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austriahttp://www.R-project.org.
Rodriguez-Galiano, F., Ghimire, B., Rogan, J., Chica-Olmo, M., Rigol-Sanchez, P., 2012. An assessment of the effectiveness of a random forest classifier for land-cover classification. ISPRS J. Photogramm. Remote Sens. 67, 93–104. https://doi.org/10.1016/j.isprsjprs.2011.11.002.
Saaty, L., 1994. How to make a decision: the analytic hierarchy process. Interfaces (Providence). INFORMS Journal on Applied Analytics 24 (6), 19–43. https://doi.org/10.1287/inte.24.6.19.
Saaty, L., Vargas, G., 2013. The analytic network process. Decis. Mak. with Anal. Netw. Process 195, 1–40. https://doi.org/10.1007/978-1-4614-7279-7_1.
Schachtschneider, K., February, C., 2013. Impact of Prosopis invasion on a keystone tree species in the Kalahari Desert. Plant Ecol. 214 (4), 597–605.
Schmidt, K., Sachse, R., Walz, A., 2016. Current role of social benefits in ecosystem service assessments. Landsc. Urban Plan. 149, 49–64. https://doi.org/10.1016/j.landurbplan.2016.01.005.
Scholes, J., Montanarella, L., Brainich, E., Barger, N., Ten Brink, B., Cantele, M., et al., 2018. IPBES: summary for policymakers of the assessment report on land degradation and restoration of the intergovernmental science-policy platform on biodiversity and ecosystem services. IPBES Secretariat. Bonn, Germany.
Shackleton, R.T., Le Maitre, C., Pasiecznik, N.M., Richardson, D.M., 2014. *Prosopis*: a global assessment of the biogeography, benefits, impacts and management of one of the world's worst woody invasive plant taxa. AOB Plants 2014 (6), plu027. https://doi.org/10.1093/aobpla/plu027.
Shackleton, R.T., Le Maitre, C., van Wilgen, W., Richardson, M., 2015a. Use of non-timber forest products from invasive alien *Prosopis* species (mesquite) and native trees in South Africa: implications for management. For. Ecosyst. 2, 1–16. https://doi.org/10.1186/s40663-015-0040-9.
Shackleton, R.T., Le Maitre, C., Van Wilgen, W., Richardson, M., 2015b. The impact of invasive alien *Prosopis* species (mesquite) on native plants in different environments in South Africa. South African J. Bot. 97, 25–31. https://doi.org/10.1016/j.sajb.2014.12.008.
Shackleton, C.M., Ruwanza, S., Sinasson Sanni, G., Bennett, S., Lacy, P., Modipa, R., Mtati, N., Sachikonye, M., Thondhlana, G., 2016. Unpacking Pandora's Box: understanding and categorising ecosystem disservices for environmental management and human wellbeing. Ecosystems 19, 587–600.
Shackleton, R.T., Le Maitre, C., van Wilgen, W., Richardson, M., 2017. Towards a national strategy to optimise the management of a widespread invasive tree (*Prosopis* species; mesquite) in South Africa. Ecosyst. Serv. 27, 1–11. https://doi.org/10.1016/j.ecoser.2016.11.022.
Shiferaw, H., Teketay, D., Nemomissa, S., Assefa, F., 2004. Some biological characteristics that foster the invasion of Prosopis juliflora (Sw.) DC. at Middle Awash Rift Valley Area, north-eastern Ethiopia. J. Arid Environ. 58, 135–154. https://doi.org/10.1016/j.jaridenv.2003.08.011.
Shiferaw, H., Schaffner, U., Bewket, W., Alamirew, T., Zeleke, G., Teketay, D., 2019a. Modelling the current fractional cover of an invasive alien plant and drivers of its invasion in a dryland ecosystem. Sci. Rep. https://doi.org/10.1038/s41598-018-36587-7.
Shiferaw, H., Bewket, W., Eckert, S., 2019b. Performances of machine learning algorithms for mapping fractional cover of an invasive plant species in a dryland ecosystem. Ecol Evol. 2019 (00), 1–13. https://doi.org/10.1002/ece3.4919.
Silvestri, S., Zaibet, L., Said, Y., Kifugo, C., 2013. Valuing ecosystem services for conservation and development purposes: a case study from Kenya. Environ. Sci. Pol. 31, 23–33.
Sutton, P., Anderson, S., Costanza, R., Kubiszewski, I., 2016. The ecological economics of land degradation: impacts on ecosystem service values. Ecol. Econ. 129, 182–192.
TEEB, 2010. The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A Synthesis of the Approach, Conclusions and Recommendations of TEEB. Progress Press, Malta 978-3-9813410-3-4 Layouted by. www.dieaktivisten.de Printed by.
Temesgen, H., Wu, W., Shi, X., Yirsaw, E., Bekele, B., Kindu, M., 2018. Variation in ecosystem service values in an agroforestry dominated landscape in Ethiopia: implications for land use and conservation policy. Sustainability 10, 11–26. https://doi.org/10.3390/su10041126.
Tessema, K., de Boer, F., Prins, H., 2016. Changes in grass plant populations and temporal soil seed bank dynamics in a semi-arid African savanna: implications for restoration. J. Environ. Manag. 182, 166–175.
Tilahun, M., Birner, R., Ilukor, J., 2014. Households' demand for mitigation of Prosopis juliflora invasion in the Afar region of Ethiopia: a contingent valuation. Manag. Prosopis Juliflora Better Pastor. Livelihoods Horn Africa Proc. Reg. Conf. May 1–May 2, 2014, Addis Ababa, Ethiop. 0568 https://doi.org/10.1080/09640568.2016.1152955.
Tolessa, T., Senbeta, F., Abebe, T., 2017. Land use/land cover analysis and ecosystem services valuation in the central highlands of Ethiopia. For. Trees Livelihoods. 26, 111–123. https://doi.org/10.1080/14728028.2016.1221780.
Tsegaye, D., Moe, R., Vedeld, P., Aynekulu, E., 2010. Land-use/cover dynamics in Northern Afar rangelands, Ethiopia. Agric. Ecosyst. Environ. 139, 174–180. https://doi.org/10.1016/j.agee.2010.07.017.
Van der Ploeg, S., De Groot, S., Wang, Y., 2010. The TEEB Valuation Database: Overview of Structure, Data and Results. Foundation for Sustainable Development, Wageningen, the Netherlands (Wageningen, the Netherlands).

van Wilgen, B.W., Forsyth, G., Le Maitre, C., Wannenburgh, A., Kotzé, F., van den Berg, E., et al., 2012. An assessment of the effectiveness of a large, national-scale invasive alien plant control strategy in South Africa. Biol. Conserv. 148, 28–38. https://doi.org/10.1016/j.biocon.2011.12.035.

Vaz, A., Kueffer, C., Kull, A., Richardson, D., Vicente, J., Kühn, I., Schröter, M., Hauck, J., Bonn, A., Honrado, J., 2017. Integrating ecosystem services and disservices: insights from plant invasions. Ecosyst. Serv. 23, 94–107. https://doi.org/10.1016/j.ecoser.2016.11.017.

Velasquez, M., Hester, T., 2013. An analysis of multi-criteria decision making methods an analysis of multi-criteria decision making methods. International Journal of Operations Research 10 (2), 56–66.

Veldman, W., Buisson, E., Durigan, G., Fernandes, W., Le Stradic, S., Mahy, G., et al., 2015a. Toward an old-growth concept for grasslands, savannas, and woodlands. Frontiers in Ecology and the Environ 13 (3), 154–162.

Veldman, W., Overbeck, E., Negreiros, D., Mahy, G., Le Stradic, S., Fernandes, W., et al., 2015b. Where tree planting and forest expansion are bad for biodiversity and ecosystem services. Bioscience 65, 1011–1018. https://doi.org/10.1093/biosci/biv118.

Viera, A., Garrett, J., 2005. Understanding interobserver agreement: the kappa statistic. Fam. Med. 37 (5), 360–363.

Vilà, M., Hulme, E., 2017. Impact of Biological Invasions on Ecosystem Services. Springer, Cham.

Vilà, M., Espinar, L., Hejda, M., Hulme, E., Jarošík, V., Maron, L., et al., 2011. Ecological impacts of invasive alien plants: a meta-analysis of their effects on species, communities and ecosystems. Ecol. Lett. 14, 702–708. https://doi.org/10.1111/j.1461-0248.2011.01628.x.

Viste, E., Korecha, D., Sorteberg, A., 2013. Recent drought and precipitation tendencies in Ethiopia. Theor. Appl. Climatol. 112, 535–551. https://doi.org/10.1007/s00704-012-0746-3.

Wise, R.M., Van Wilgen, B.W., Le Maitre, D.C., 2012. Costs, benefits and management options for an invasive alien tree species: the case of mesquite in the Northern Cape, South Africa. J. Arid Environ. 84, 80–90.

Woody Biomass, 2000. Land Use and Land Cover Mapping of the Afar Region. Ministry of Agriculture, Addis Ababa.

Yosef, T., Mengistu, U., Solomon, A., Mohammed, K., Kefelegn, K., 2013. Camel and cattle population dynamics and livelihood diversification as a response to climate change in pastoral areas of Ethiopia. Livest. Res. Rural. Dev. 25 (9), 1–10.

Zewdie, W., Csaplovics, T., 2015. Remote Sensing based multi-temporal land cover classification and change detection in northwestern Ethiopia. European Journal of Remote Sensing 48, 121–139. https://doi.org/10.5721/EuJRS20154808.