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ORIGINAL PAPER

Assessing the impacts of watershed interventions using ground data and remote sensing: a case study in Ethiopia

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Abstract

Quantifying the temporal and spatial changes due to watershed interventions is important for assessing the effectiveness of natural resource management practices on vegetative cover and sediment management. This study assessed the performance of natural resource management in a target site (Aba Gerima) and compared the collateral impacts on neighbouring watersheds in Ethiopia in terms of land-use land-cover change. Changes in the extent of cropland, grassland and shrubland were assessed in the target watershed and the non-treated neighbouring watersheds using temporal satellite imagery. In addition, ground monitoring was applied to quantify the impacts on sediment accumulation, fodder biomass and vegetative cover intensity. The study findings showed substantial changes over the study period: mainly, a change from degraded and barren land to restored vegetation in the target watershed, but a continued trend of land-use change from perennial vegetation to cropland in the neighbouring untreated watersheds. There was a decrease in the rate of conversion of vegetative land cover to cropland in the target watershed, and significant on-site changes in sediment retention, fodder productivity and vegetation intensity. The study findings demonstrate a link between management interventions and improvement in soil and vegetation ecosystem functions. These results not only indicate that watershed-level interventions improve on-site soil and water environmental services but also underline the role of community managed land-use regulations in reducing pressure on natural land-use systems and thereby serve the major goal of up-scaling sustainable land management.

Keywords Agriculture croplands · Land-cover changes · Natural resource management · Restored vegetation · Satellite imagery

Introduction

Subsistence food production has been practiced for millennia in the Ethiopian highlands. However, land degradation due to soil erosion is now seriously threatening the sustainability of agriculture in the region (Providoli et al. 2019). Over 85 percent of the land is reported to have been degraded, the annual cost of which is estimated at $4.3 billion annually (Gebreselassie et al. 2016). The loss of fertile topsoil threatens agricultural productivity, and the heightened risks of sedimentation of water bodies require calls for efforts to halt erosion and restore these valuable landscapes (Abera et al. 2020). Given this context, an understanding of the widespread success of land restoration efforts, and their possible impacts on the on-site environmental services and neighbouring areas as well, is necessary to inform the management of soil and water resources in valuable agricultural landscapes such as these.

In Ethiopia, as in other locations in the world, several studies have reported on-site impacts of natural resource management (NRM) on soil loss, sediment yield, discharge, land cover and productivity within catchment boundaries (Adimassu et al. 2016; Ebabu et al. 2019; Kassawmar et al.

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2018a; Tilahun et al. 2012). For instance, interventions that improve soil and water conservation, such as stone or soil bunds, trenches, vegetative hedgerows, contour farming, cover crops and area closure, have been particularly successful in Ethiopia. (Klik et al. 2018) illustrated that stone bunds increase infiltration time, leading to an increase in the soil water content of the agricultural lands. (Ebabu et al. 2019) reported significant reductions in run-off and soil loss upon adoption of improved land and water management interventions. A combination of stone bunds and trenches on degraded non-crop land improved soil water storage and vegetation cover (Wolka et al. 2018).

One of the key objectives of soil erosion management is to maintain soil protection and reduce disturbance, thereby minimizing vulnerability to erosion. Vegetation restoration is one of the best watershed interventions for improving protective capacity (land cover) and fighting soil erosion (Mekuria 2013; Mekuria et al. 2009). Ongoing NRM programmes have brought about improvements in soil moisture and groundwater recharge, thus contributing to an increase in vegetation cover (Desta et al. 2019; Rashid et al. 2011; Wolka et al. 2018). If carried out over a large area, NRM practices can result in major land-use changes beyond the target watershed boundary (Amede et al. 2020). Vegetation restoration can have a significant effect on intensively grazed lands, which are more vulnerable to soil loss due to removal of its protective vegetation cover and increased run-off induced by livestock compaction of the surface soil (Ebabu et al. 2019; Mwendera and Saleem 1997). Toward restoring degraded ecosystems, area exclusion (i.e. areas closed to human and animal interference) can contribute to control of soil erosion, improvement of nutrient cycling and biomass production (Mekuria 2013; Mekuria et al. 2009). Area closure management and protection have proved to be effective in restoring degraded steep slopes and hills, earning positive feedback from the local communities (Descheemaeker et al. 2010, 2006a, 2006b; Mekuria 2013). This makes it a promising option in other areas as well. Soil conservation strategies using restorative measures of soil and cover management are therefore essential to prevent soil degradation (Di Prima et al. 2018). Thus, investments in NRM practices are vital for rural development in Ethiopia (Kassawmar et al. 2018b; Kato et al. 2019). However, interpreting changes in land use and cover intensity requires monitoring of the temporal and spatial effects of NRM practices within the target watershed and beyond.

Numerous studies have assessed watersheds using different methods and approaches (Alemayehu et al. 2009; Atoma et al. 2020; Gashaw et al. 2018; Sewnet 2016; Tadesse et al. 2017). Alemayehu et al. (2009) assessed integrated watershed management (IWSM) in upper Agula watershed in Ethiopia and found significant modification and conversion of land use and cover of the watershed over four decades (1965–2005). These researchers noted decreased soil erosion, increased soil moisture, reduced sedimentation and run-off. Gashaw et al. (2018) conducted a study in the Geleda watershed in the Blue Nile basin in the northwestern highlands of Ethiopia to measure erosion rates and to map erosion risks for prioritization of conservation measures. They reported that expansion of cultivated land had resulted in high soil losses mainly in the steep slope areas of the watershed. Tadesse et al. (2017) estimated the spatiotemporal changes in the land-use/land-cover pattern and soil erosion in the Yezat watershed in Ethiopia mainly using NDVI and Revised Universal Soil Loss Equation and found changes in the LULC and a decrease in soil erosion. Atoma et al. (2020) studied LULC changes and their potential impacts on soil erosion in the Huluka watershed in Oromia regional state of Ethiopia during the period 1998–2018. Integrating satellite data and socio-economic data, Sewnet (2016) analysed LULC changes in the Infraz watershed and found an increase in inappropriate agricultural activities.

While there are such numerous research reports on the on-site impacts of watershed interventions, there have been few studies attempting to relate the results of NRM interventions in a 'treated' watershed to their impact on neighbouring 'untreated' watersheds. The relation between the on-site successes of an intervention and its positive impacts in neighbouring watersheds has been noted anecdotally by Kassawmar et al. (2018b) but rarely studied. There is a real need to interpret such site-specific and spatial improvement of water and soil in a surrounding watershed in terms of land-use change and vegetation cover intensity. However, quantifying and monitoring such impacts presents a major challenge to scientists, watershed managers and policymakers. This is due to a lack of necessary data on several economic, social and environmental indicators and associated driving factors. In many cases, from a statistical point of view, it is difficult to conduct a comparative analysis of the impacts in a target watershed and related impacts in a neighbouring watershed as there is the challenge of heterogeneity. Lacking the data necessary to assess the on-site impact of an intervention on its target watershed as well as its effect on neighbouring watersheds, a more valid approach would be to determine the relative change of impacts over a number of years using proxy indicators. One cost-effective, rapid and scalable option is that assessment of watershed interventions by combining field monitoring and remote sensing tools. Given this context, we attempted to quantitatively assess whether the successes of a targeted NRM interventions in a treated watershed could achieve watershed-wide environmental impacts, and whether it was possible to detect linkages via flow-on effects on LULC in the adjacent neighbouring watersheds. Here we aimed at quantifying the performances and resulting on-site effects of NRM practices in the target watershed and the relative changes in land use and land cover in the neighbouring watersheds.

Thus, the objectives of our study were: (a) quantifying the extent and distribution of NRM interventions; (b) assessing the impacts of watershed interventions in a treated model watershed; (c) estimating the relative land-use and land-cover changes in a treated model watershed and untreated neighbouring watersheds. The research was carried out for Aba Gerima watershed (treated watershed) and its non-treated neighbouring watersheds during pre- and post-intervention years (2002, 2013, and 2019).

Table 1 Geographical features of watersheds in the study area

Code	Name	Area (ha)	Stream length (km)	Elevation (m)	
				Maximum	Minimum
WS1	Abe Gerima	893	74	2121	1889
WS2	Zigba	954	77	2045	1831
WS3	Yedemo/Robit	1206	97	1969	1812
WS4	Neber Wenz	799	62	2247	1927
WS5	Laguna	2067	164	2251	1850

Materials and method

Description of study area

The Aba Gerima watershed (WS1) lies in the central part of Amhara region of north western Ethiopia in the Lake Tana sub-basin (Fig. 1a). The total geographical area of the study landscape is 5,919 ha (Table 1) including the four neighbouring watersheds bordering the Aba Gerima watershed (Fig. 1). This is a humid climatic region with mean annual rainfall ranging from 900 to 1200 mm. The main rainy season begins in May and ends in September with peak rainfall occurring in July and August. The upper and lower parts of the watershed receive variable rainfall.

Agriculture is the mainstay of livelihoods in this part of Ethiopia. It is dominated by livestock raising and cereal cultivation which occupies the greatest area (~60%) in the watershed. Livestock production although occupying less area is crucial for livelihoods, home gardens, fruit and khat (*Catha edulis*) production served by shallow hand dug wells are common in villages located along the streams and rivers.

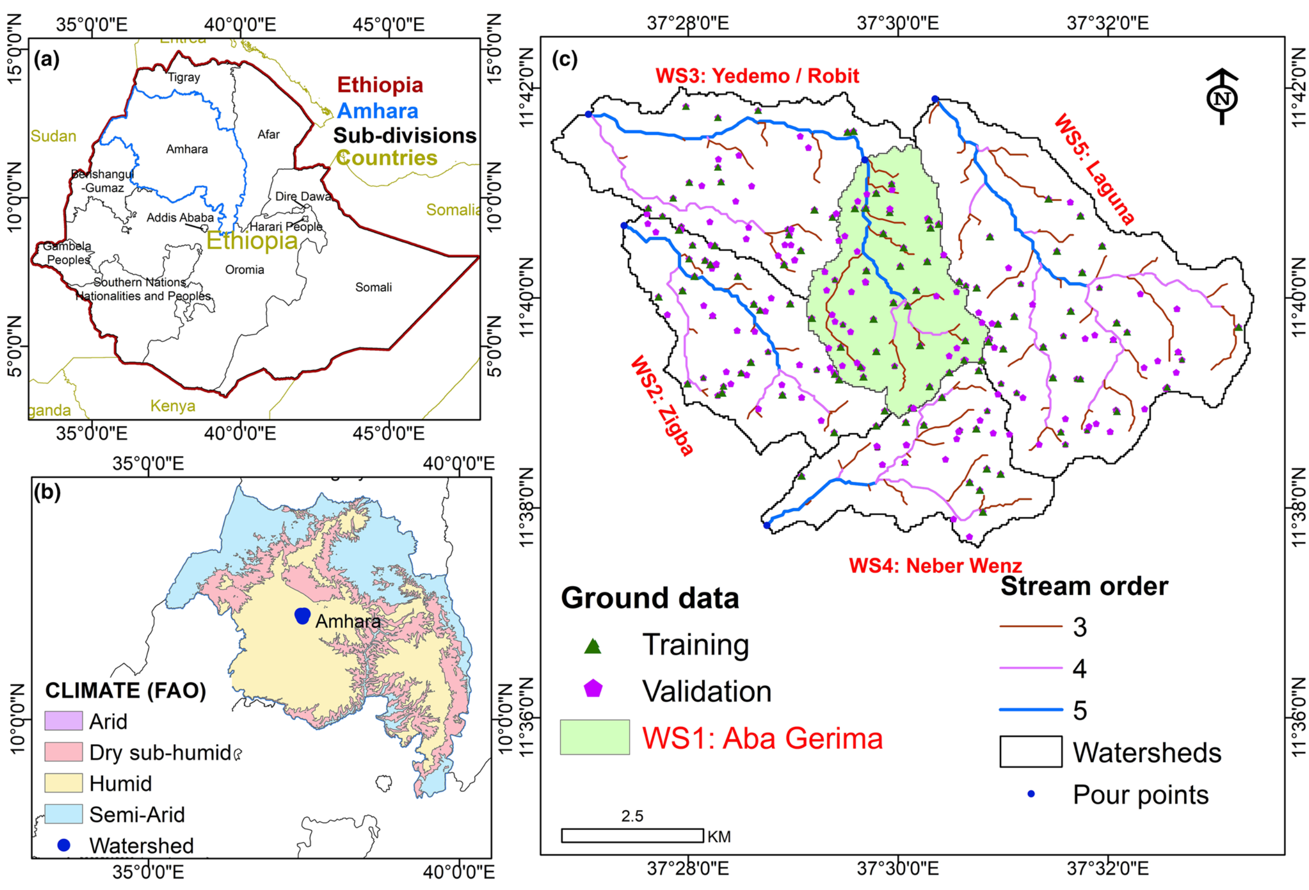

Fig. 1 Location of study watersheds **a** in the central part of Ethiopia in the study area; **b** climate distribution; and **c** Study watersheds

In 2012, the Aba Gerima watershed (983 ha) was established by the partnership of local implementing actors such as agricultural offices, research institutes and land users to serve as a learning site to demonstrate best agricultural and NRM practices and to facilitate learnings in order to scale up the impacts (Desta et al. 2019). The watershed learning initiative implemented different interventions for close to seven years, 2012–2018, using free labour mobilized from the community. The watershed is dominated by arable land use and patches of non-croplands. The watershed interventions incorporated different NRM practices such as farm bunds, vegetative hedgerows and run-off waterways on cultivated lands; check dams on gullies; and exclosures and in situ moisture harvesting structures to rehabilitate degraded hills (Benson et al. 2013; Desta et al. 2019; Kato et al. 2019). In addition, agricultural practices such as improved crop varieties, fodder species, livestock breeds, agricultural machinery and intensified home garden activities were demonstrated and promoted (Benson et al. 2013; Desta et al. 2019; Kato et al. 2019).

Data sources

Satellite imagery

To analyse the pre-intervention situation and post-intervention impacts of integrated NRM practices, we selected high-resolution spatiotemporal satellite images from three years, 2002, 2013 and 2019, on the basis of availability of cloud-free imagery: Landsat for the year 2002 and 2013 (source: http://edcsns17.cr.usgs.gov/NewEarthExplorer/) and Sentinel-2 for 2019 (https://scihub.copernicus.eu/) (Table 2). In addition, imagery from Landsat (30 m, 16-day cloud free) and Sentinel-2 (10 m 12-day) was downloaded from the cloud computing GEE for three study years. The Normalized Difference Vegetation Index (NDVI) was calculated using Sentinel bands and further processed to create monthly NDVI maximum value composites (NDVI MVC) for each year (Gumma et al. 2011a). Shuttle Radar Topography Mission (SRTM) data were downloaded from USGS Earth Explorer (1-Arc Second) to generate digital elevation model at 30 m spatial resolution.

Ground survey data

In addition, ground data were collected for the years 2002, 2013 and 2019 to assess the pre- and post-intervention situation in the watersheds. The collection points targeted the major LULC types distributed across the study area. Ground data were collected from available Google Earth imagery and validated against the Landsat NDVI time series and the allocated ground points for 2002 and 2013. For the year 2002, data from 200 points were collected, out of which 74 points were used for training and 126 points for validation. For the year 2013, data were collected from 197 points, out of which 73 points were used for training and 124 points for validation. For the year 2019, there were 191 points, out of which 73 points were used for training and 118 points for validation. At each location, data were collected from 90 m × 90 m plots with GPS locations and land-use category. Samples were obtained from within large contiguous areas of a particular land-use category.

Field data measurement

To assess the impacts of watershed interventions, particularly the extent of rehabilitation practices within the target watershed, annual field monitoring was conducted on soil conservation bunds and sediment management induced by the bunds during the period 2012–2018. In addition to the primary data sources, secondary sources were used to substantiate the performance of interventions and associated impacts.

Methods

Our approach to assessing the impact of NRM practices in the target watershed and the neighbouring watersheds involved an analysis of the temporal LULC changes as well as examining the performance of watershed interventions and their resulting changes in terms of fodder biomass productivity, vegetation cover and sediment management in the model watershed (Fig. 2).

Table 2 Characteristics of image data used to monitor changes in the watersheds

Satellite data	Spectral bands #	Wavelength (μm)	Application
Landsat-8 data sets	Band 4 (Red)	0.64 to 0.67	Discriminates vegetation slopes
	Band 5	0.85 to 0.88	Emphasizes biomass content and shorelines
SRTM 30 m	C-band radar	56,000	Slope
Sentinel-2	Band 4 (Red)	0.65 to 0.68	Vegetation classification
	Band 8 (NIR)	0.78 to 0.90	Sensitive to biomass and chlorophyll

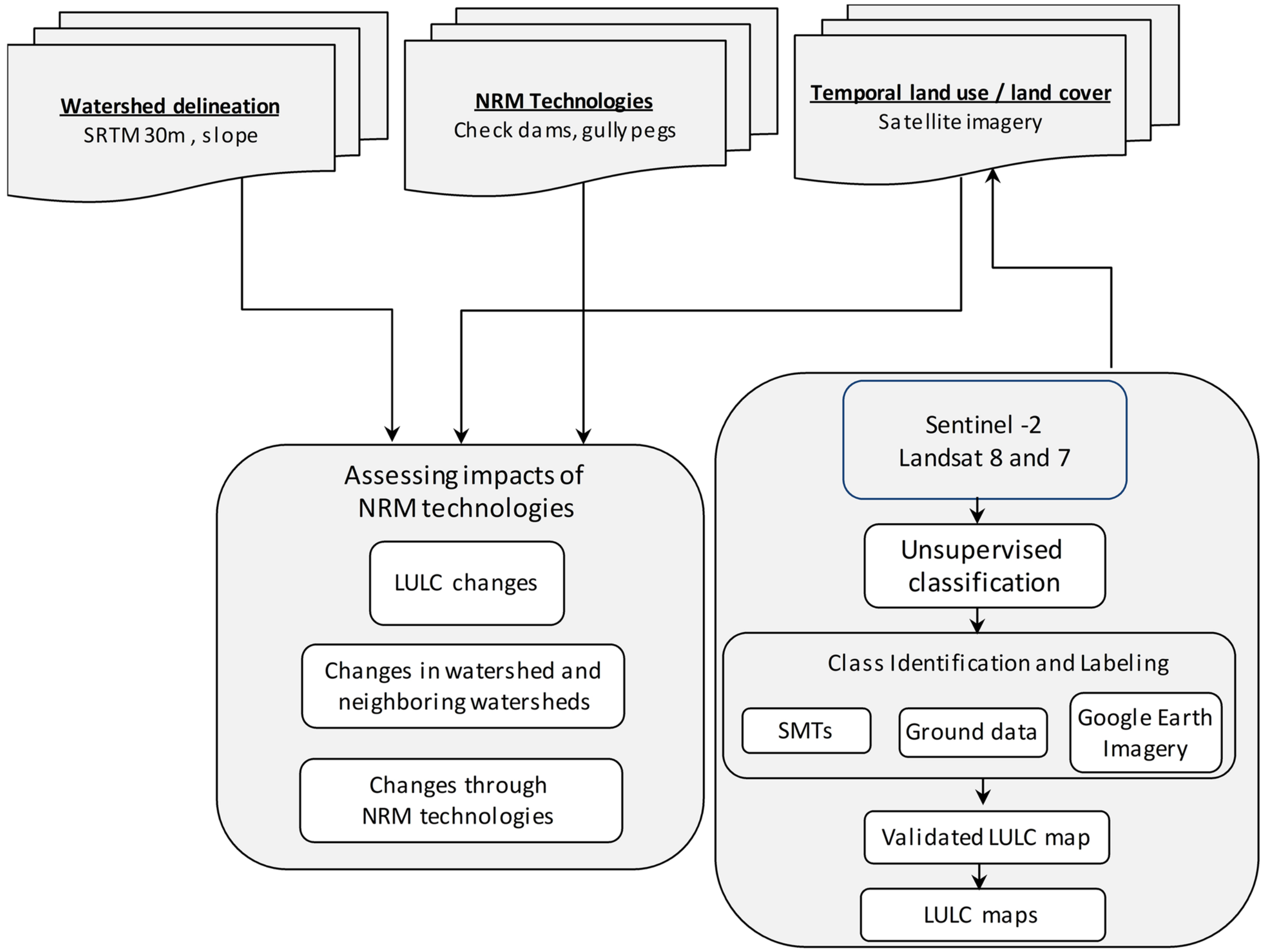

Fig. 2 Methods and approaches to analysing the impact of NRM technologies

Land-use/land-cover analysis

In order to assess the pre- and post-intervention situation, the LULC changes occurring over 17 years (from 2002 to 2019 through 2013) were mapped following the process described in detail by (Gumma et al. 2011c; Thenkabail et al. 2007). Briefly, LULC areas were mapped using Landsat-7, Landsat-8 and Sentinel-2 time series data and temporal profiles supported by ground data. The process begins with downloading the NDVI maximum value composites (MVCs) of Landsat and Sentinel-2 time series data and stacking them into a single data set for 2002 (8 layers), 2013 (10 layers) and 2019 (12 images). Each year's stacked image was classified using the ISOCLASS cluster isodata classification, with 50 classes, 50 maximum iterations and a convergence threshold of 0.99. In the study areas, some of the features do not have training data, where unsupervised classification was used to identify different LULC types. Simultaneously, mean spectral values were generated using a signature set option for all 50 classes. Class identification was carried out on the basis of temporal profiles and Google Earth high-resolution imagery. The classes obtained from the unsupervised classification were combined into six classes and titled on basis of spectral similarity with a magnitude of vegetation index, intensive ground data information and Google Earth high-resolution imagery. Using spectral matching techniques, the classes were related for all the years (Gumma et al. 2011c; Thenkabail et al. 2007).

Using NDVI time-series plots, changes in the LULC area were mapped using spectral matching techniques (Gumma et al. 2011c; Thenkabail et al. 2005). The NDVI data were further processed to create NDVI MVC for each month of wet season using Eq. 1:

$$\mathrm{MVC}_i = \mathrm{Max}(\mathrm{NDVI}_{i1}, \mathrm{NDVI}_{i2}, \ldots) \quad (1)$$

where MVC_i is the monthly maximum value composite of the ith month (e.g. "i" is January to December). $i1$, $i2$ and so on are images of every month.

Accuracy assessment of the temporal LULC classifications was carried out using validation data, which was collected over three years from different years. Figure 1c shows the distribution of the training and validation samples. Validation data are independent of the reference training data and are reserved for validation purpose. Accuracy assessment was performed using error matrices (Congalton and Kass 2008).

The LULC patterns (cropland, grassland and shrubland) for the years 2002, 2013 and 2019 were compared. The change in cropland area was identified using spectral matching techniques i.e. from cropland class to another class (Gumma et al. 2011b; Thenkabail et al. 2007) by taking into consideration of duration, magnitude and the peak of NDVI curve. Change from 2002 to 2013 and on to 2019 was estimated LULC class-wise using Eq. 2.

$$\mathrm{CD}_{ij} = \left(\mathrm{LULC}_i \times 10\right) + \mathrm{LULC}_j \tag{2}$$

where CD_{ij} is the change detected, $LULC_i$is LULC for the ith year and $LULC_j$ is LULC for the jth year.

Leaf area index

The leaf area index (LAI) was generated as per Eq. 3 using Landsat time series data with soil-adjusted vegetation index (SAVI) computed using Eq. 4 (Huete 1988; Qi et al. 1994; Reyes-González et al. 2019; Rondeaux et al. 1996; Thenkabail et al. 2000). In this study, we used the spectral response of leaves which is unique compared to that of other parts of the plant.

$$\mathrm{LAI} = \frac{-\ln\left(\frac{0.69-\mathrm{SAVI}}{0.59}\right)}{0.91} \tag{3}$$

For Landsat images used in this study, SAVI was computed using the formula:

$$\mathrm{SAVI} = \frac{(1+L)(B5-B4)}{L+B5+B4} \tag{4}$$

where L is a soil factor, taken to be 0.1, B5 is the spectral reflectance in Near Infrared (NIR) and B4 is spectral reflectance in Red.

Assessing soil and vegetation impact of NRM practices

The impact of soil and water conservation practices and land use and vegetation management on sediment management, fodder biomass and vegetation cover were assessed and quantified at the model watershed scale. In general, the authors adopted multi-stage monitoring techniques to assess the impacts of watershed interventions using simple and rapid methods and tools. The field data collection methods are summarized briefly below. The details can be found in a research report by Desta et al. (2019).

Quantifying sediment retained on bunds To quantify the sediment management of a treated watershed, a mapping unit approach of stratifying data sampling into homogeneous categories of slope, soil and land use and cover features was employed. Once the homogeneous mapping units (HMU) were identified and delineated, four steps were considered to estimate the total sediment retained by the soil conservation bunds:

1. Calculate the density of soil conservation bunds (i.e. length of bunds per unit area) at each mapping unit using on-screen digitization of bunds from high-resolution Google Earth images;
2. Measure the cross-sectional area of sediment retained on bunds using sample bunds in the HMUs. Each HMU was further classified into upper, middle and lower slope positions to account for the topo-sequence effects of soil erosion. In each slope position, three successive (namely upper, middle and lower) soil conservation bunds were selected. In each of the three successive bunds, 30 m length of bund was taken for sampling. Bunds were constructed with stone and/or soil embankment (lower riser) and a sediment retaining basin or trench on the upper side. The width of the basin area of bunds was 60 cm, and the depth varied from 30 to 40 cm depending upon the slope and soil depth conditions. Thus, the accumulated depth of retained sediment over years was measured along 30-m basin area/trench section at 6-m intervals (5 sampling points) using installed 6-cm-diameter wooden pegs.
3. The total volume of sediment retained in a treated watershed was quantified by multiplying the depth and width of sediment retained on the bunds per unit length by the corresponding total length of bunds in each HMU; and
4. The rate of sediment management after NRM interventions was compared with the rate before intervention or the baseline rates of soil loss and sediment yield.

Assessing fodder biomass and vegetation cover Planting of fodder shrubs and grasses is one of the interventions employed to rehabilitate degraded lands and to integrate shrubs with the physical soil conservation bunds. In order to quantify the intervention impact on fodder availability in crop lands, biomass of fodder species planted on soil conservation bunds was sampled in each HMU using the same sampling procedure used for sediment, except that the fodder biomass harvest samples were taken from 5 m length of bund. For the remaining fodder niches such as area exclosures and rehabilitated gullies, 5 m by 5 m sampling plots were used for biomass harvest estimation. To complement the biomass measurement and assess the change in vegetation cover in a treated watershed, temporal canopy cover analysis was performed over selected intervention periods using Google Earth images. The change analysis was made through the spatially explicit HMUs.

Results and discussion

Extent and distribution of NRM practices

Sediment management practices such as cross-slope erosion control structures and vegetation cover management were implemented to mitigate land degradation and control soil erosion. Using Google Earth images, the spatial extent and distribution of soil conservation bunds on crop lands and vegetation cover management options such as fodder plantations and degraded area exclosures were mapped in each HMU. The results indicated rehabilitation of more than 900 ha of crop lands including annual maintenance, 62 ha of degraded lands and 15 ha of gully areas (Table 3). Figure 3 (left map) illustrates the spatial density of soil conservation bunds constructed over seven years (2012–2018). The community-led watershed intervention constructed a total of 127 km of soil bunds and stone terraces on crop land in the target watershed with an average density of 217 m of bunds

Table 3 Type and coverage of NRM interventions in Aba Gerima watershed (source: Desta et al. 2019)

Interventions	Area in ha						
	2012	2013	2014	2015	2016	2017	Total
Terrace structures on cultivated land	341.3	228.3	180.5	78	89	20	937
Check dams on treated gully	0	4	9	0	2	0	15
Rehabilitated hills	20	6	20	12	3	0.5	61.5
Vegetative measures on cultivated land, gully and degraded hills	361.3	238.3	209.5	162.0	295.5	191.5	1458

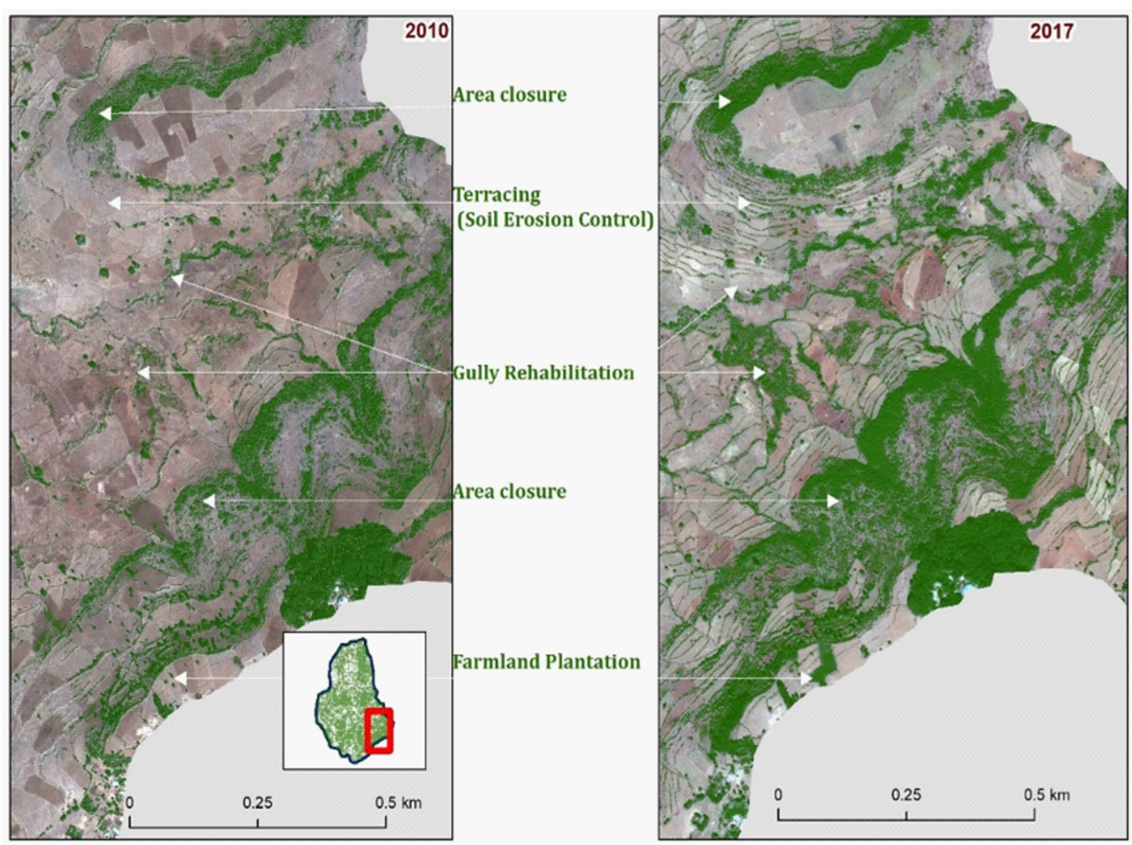

Fig. 3 Intensity of vegetation cover management practices such as gully management, exclosure and fodder plantation on bunds before (left) and after (right) watershed interventions

Table 4 Fodder biomass productivity at different land-use areas, bund density and sediment retained on bunds in the Aba Gerima watershed

Fodder productivity at different land uses (t ha^{-1})			
Area closure on degraded lands	Grass land	Gully area	Farm bunds (kg per 5 m)
5–11	16–24	8–22	10–40
Bund density and sediment retained on bunds			
Bund length (km)	Density of bunds (m ha^{-1})	Sediment retained on bunds (m^{3}m^{-1})	Sediment retained m^3 (tonnes)
126.8 ± 10.7	216.8 ± 85.1	0.057 ± 0.016	7478 ± 757 (8973 ± 909)

per hectare (i.e. varying between 100 m ha^{-1} and 330 m ha^{-1} in different sub-catchments, and from 10 ha^{-1} to 900 m ha^{-1} over the entire watershed). The variation was attributed to slope and land-use differences. The lower slopes received a smaller number of bunds. Among other watershed interventions, vegetation cover management practices including gully rehabilitation, exclosure of degraded hills and integration of fodder shrubs and grasses on soil conservation bunds were implemented in different niches (Fig. 3). As seen in Fig. 3, niches that were treated with fodder plantations, woodlots and exclosure of degraded hills and overgrazed pastureland show a clear improvement in the intensity of vegetation cover and greenness in 2017 compared to the 2010 Google Earth image. Eventually, as reported by Kato et al.(2019), NRM practices showed higher adoption rates in the target watershed than in the nearby control watersheds (Table 4). In particular, higher adoption rates were seen for soil bunds and stone terraces. From these results, it can be concluded that the NRM practices promoted in the showcase watershed were effective in increasing adoption and use of sustainable land management practices and investments.

Change in sediment management by soil conservation bunds

Figure 4 presents the average amount of cumulative sediment retained on conservation bunds in the treated watershed and its association with spatial change in LULC. The average

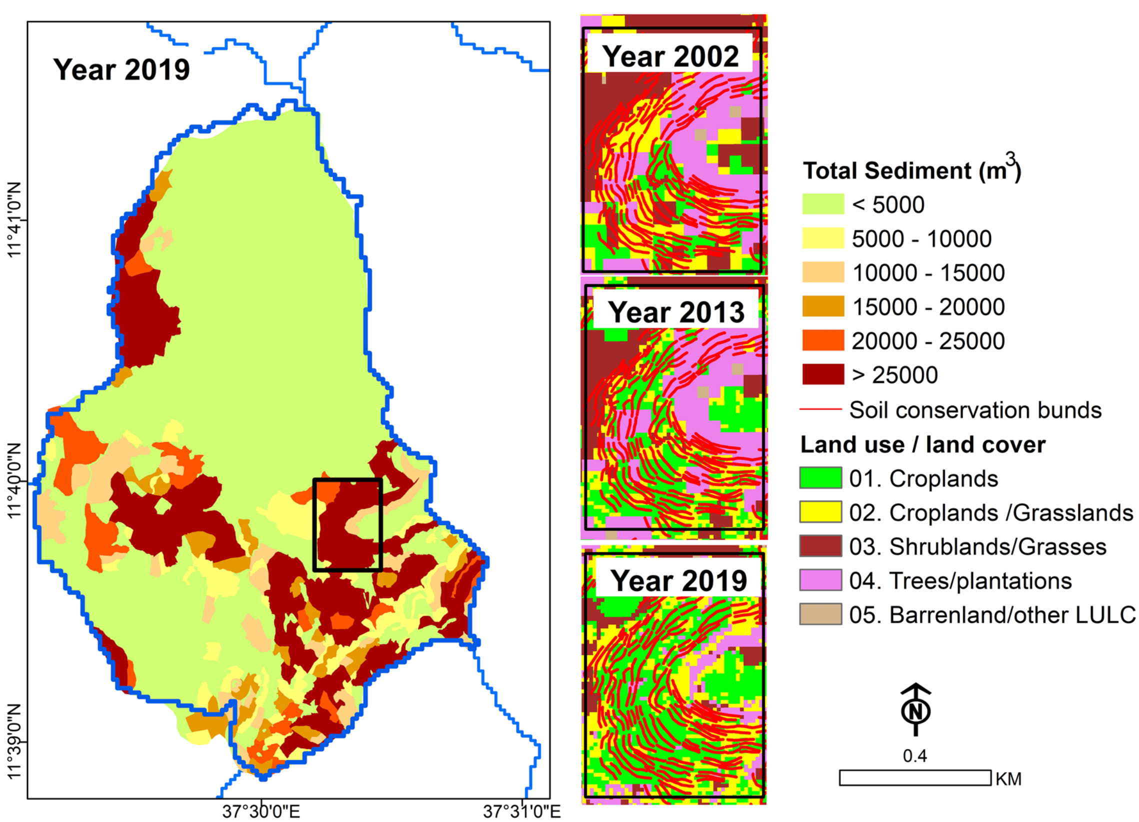

Fig. 4 Sediment retention in soil conservation bunds (left) and changes in LULC (right) attributed to sediment storage

sediment retention was 0.057m^3m^{-1}, varying between 0.027 m^3 m^{-1} and 0.081m^3m^{-1} in different sub-catchments with different slope, soil and land-use characteristics and corresponding bund density (see Fig. 4). Combining soil and/or stone bunds with fodder or grass vegetative hedgerows increases the storage capacity in the range of 0.025–0.075 m^3 m^{-1}. Elephant grass and Sesbania sesban hedgerows provided higher sediment storage efficiency (0.06–0.075 m^3m^{-1}). Despite the fact that the design of soil conservation bunds was more or less uniform, there was relatively greater sediment accumulation on bunds located in the upper slope positions (Fig. 4). This implies that sediment delivery is decreasing downslope along the topo-sequence. This is attributed to the scale effect of landscape structures—bunds, exclosures and gully rehabilitation. By implementing a total of 127 km of bunds in the watershed (Sect. 3.1) with average sediment retention of 0.057 m^3m^{-1}, about 8973 tonnes of sediment is estimated to have been retained that otherwise have been washed into the rivers out of the watershed. This is equivalent to a 9.9 t ha^{-1} rate of sedimentation in the watershed which is below the tolerable rate of erosion, 10 t ha^{-1}. This demonstrates a significant amount of soil erosion reduction as compared to the current average erosion rate (25 to 65.0 t ha^{-1}) in the Lake Tana basin (Setegn et al. 2008). Apart from the improved soil and water environmental services by reducing sediment yields, effective sediment retention of bunds has led to increasing fodder and crop productivity. Long-term crop yield monitoring on the conserved lands revealed a 30–40% increase in crop yield due to sediment retention on the conservation bunds (Hurni 2000). About 55–75% sediment deposition on the upper part of bunds contributed to a significant crop yield increase as reported in Anjeni (Subhatu et al. 2018).

Fodder biomass production

Vegetative fodder species such as *Sesbania sesban*, pigeonpea and napier grass were planted in hedgerows on conservation bunds in the Aba Gerima watershed (see Figs. 3 and 7). Sample data showed that the fodder legumes and local grass species harvested on the bunds added sufficient fodder biomass to feed livestock (Table 4). Under farmers' field management condition, potentially two harvestings of fodder shrub species per year are possible. A one-time harvest produced roughly 8–15 kg and 20–40 kg of average fresh fodder and grass biomass per 5 m length of soil bund under poor and good production conditions, respectively (Table 4). This amount of fodder was extra feed over the basal grazing practice. This indicates that hedgerows on one hectare of soil conservation bunds can easily accommodate 2100–2800 kg of fodder shrubs which can feed a cow throughout the year (6 kg of fresh fodder per day) with a supplementation of basal grazing. Other researches too have reported increased fodder productivity and feed quality (Mekoya et al. 2008; Sisay and Mekonnen 2013). An additional benefit, nearly 2.8 and 0.7 t ha^{-1} $year^{-1}$ of dried forage, was obtained from local grass and elephant grass combined with soil bunds (Amare et al. 2014). In addition to farm bunds, rehabilitated gullies are another niche with the potential to produce feed for livestock. Based on field fodder biomass measurement, the amount of fodder produced in this niche was on average 8–20 t ha^{-1}. This suggests that gully beds and sides can be converted into productive land to generate feed for livestock. Farmers also harvested large amounts of pasture on area exclosures on degraded lands and pasture lands. This produced an average of 5–10 t ha^{-1} and 15–25 t ha^{-1} of fresh grass biomass from area exclosures and grass lands, respectively (Table 4). Added fodder biomass production thus aided diversification of farmers' income sources. Also, by producing sufficient fodder to support livestock communities came to understand the benefits of changing their production systems (Kato et al. 2019). In addition to the fodder benefits to the farmers, elimination of grazing pressure and re-vegetation of the environment occurred rapidly in the moist soil held by the soil bunds, check dams and area exclosures (Desta et al. 2019).

Spatial distribution of LULC pattern in model and neighbouring watersheds

Figure 5 and Table 5 show the LULC areas for the years 2002, 2013 and 2019. The main observation is that in WS1 there was a consistent increase in cropping area. From 2002 to 2019, cropland in this watershed increased by 32%, and other cropland classes mixed with grasslands increased by 140%. There was on the other hand a decrease in shrublands, tree-shrub mixed lands and barren lands by 45%, 34% and 75%, respectively.

In the neighbouring (untreated) watersheds, the extent of cropland increased by 46%, 3%, 70% and 66% for Zigba, Yedemo Neber Wenz and Laguna watersheds, respectively (Table 5 and Fig. 5). There was an appreciable change in the extent of cropland in the entire landscape from 1641 to 2254 ha (37%). In the case of cropland mixed with grassland, the increase was from 732 to 1743 ha (138%). There was a decrease in barren and shrublands, which reflects the conversion of such lands into managed grasslands. The conversion of barren land to vegetative uses was due to watershed interventions.

Independently, accuracy assessment was performed using ground data for the years (2002, 2013 and 2019). Overall accuracy varied from 83 to 88% for different years. The accuracy rates for various LULC classes are provided in Appendix 1.

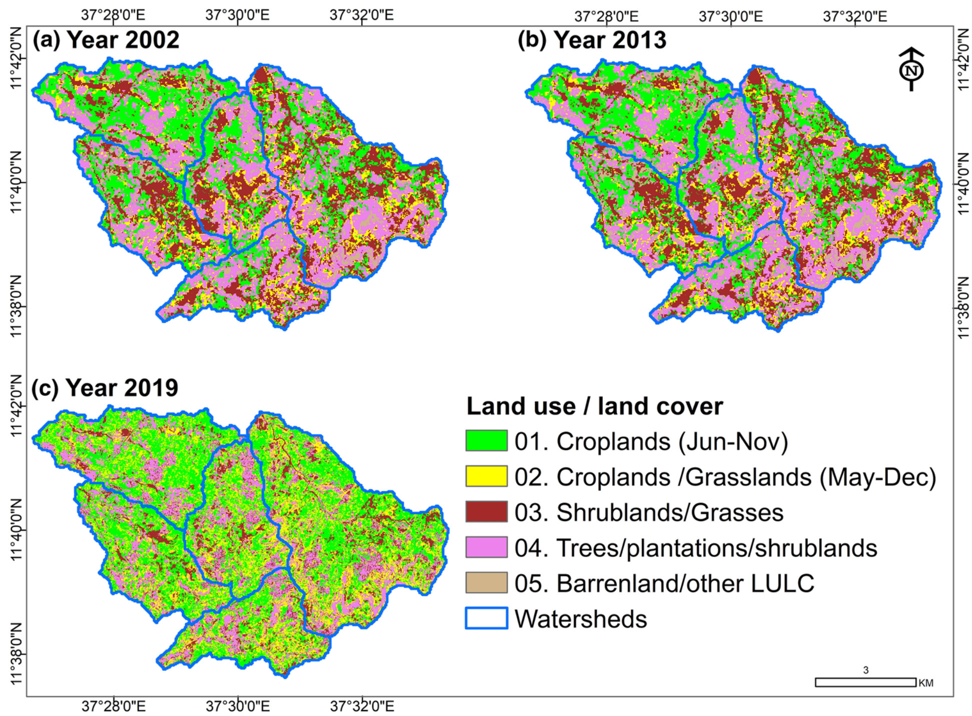

Fig. 5 Spatial extent of LULC in the study watersheds in 2002, 2013 and 2019

LULC changes in model and neighbouring watersheds

Figure 6 and Table 6 illustrate the conversion of non-croplands into croplands. In the study watershed WS1, nearly 303 ha of non-croplands were converted to cropland from 2002 to 2013. This was mainly due to an increase in vegetation/trees as a result of improved soil fertility and rainwater management, as can be seen from the increase in LAI (Table 7 and Fig. 8) for the corresponding years. During the

Table 5 Land-use/land-cover areas for different years extracted from the present study

Year	Watershed name	Area (ha)				
		01. Croplands (Jun–Nov)	02. Croplands/grasslands (May–Dec)	03. Shrublands/grasses	04. Trees/plantations/shrublands	05. Barren land/other LULC
Year 2019	WS1: Aba Gerima	299	274	111	197	12
	WS2: Zigba	398	257	113	172	14
	WS3: Yedemo	558	298	93	234	23
	WS4: Neber Wenz	253	251	101	185	10
	WS5: Laguna	747	662	285	361	13
	Total area	2254	1743	701	1149	72
Year 2013	WS1: Aba Gerima	242	158	176	288	30
	WS2: Zigba	345	159	188	235	26
	WS3: Yedemo	535	227	140	264	40
	WS4: Neber Wenz	195	123	162	286	33
	WS5: Laguna	541	343	485	614	83
	Total area	1858	1010	1152	1687	213
Year 2002	WS1: Aba Gerima	227	114	204	298	49
	WS2: Zigba	273	115	231	245	89
	WS3: Yedemo	542	88	162	274	139
	WS4: Neber Wenz	149	120	182	310	38
	WS5: Laguna	449	295	555	662	106
	Total area	1641	732	1334	1791	421

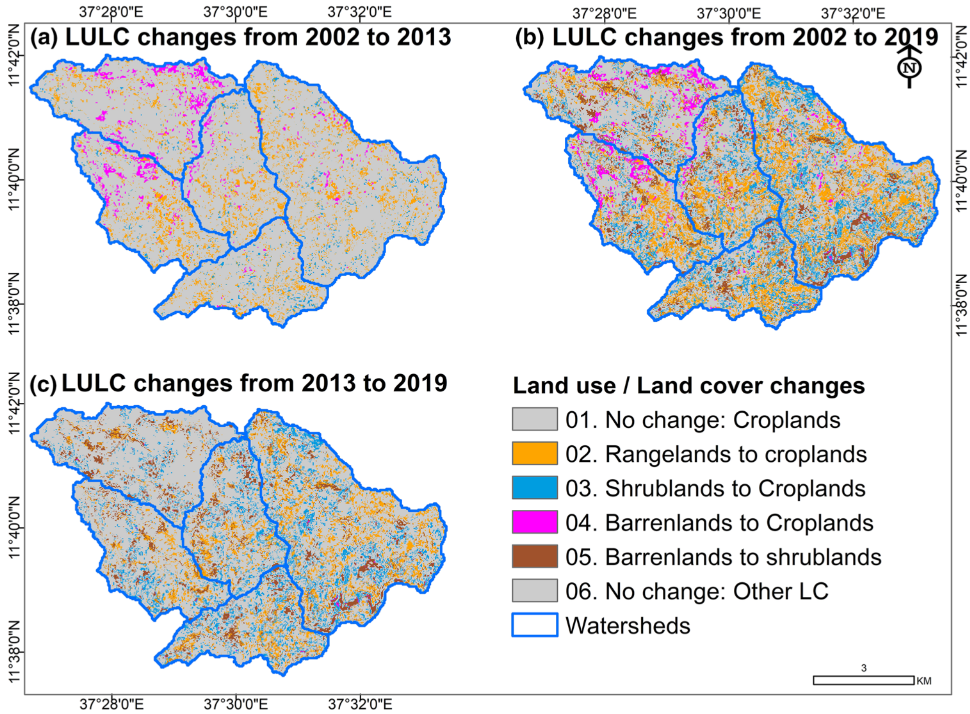

Fig. 6 Spatial extent of LULC changes in the study watersheds from **a** 2002 to 2013; **b** 2002 to 2019 and **c** 2013 to 2019

post-intervention period in WS1 (2013 to 2019), 187 ha of non-cropland were converted into cropland. Major changes were noticed during 2002–2013, implying a decrease in the rate of conversion of non-crop land to cropland. Previously cultivated lands that had been abandoned due to degradation were now being restored by terracing for crop use (see overlap of terraces and land-cover layers in Fig. 7). In the neighbouring four watersheds, nearly 932 ha of non-cropland

Table 6 Land-use/land-cover changes in three phases (from 2002 to 2013; from 2013 to 2019; and from 2002 to 2019)

Year	Watershed name	Area in ha					
		01. No change: Croplands	02. Rangelands to croplands	03. Shrublands to Croplands	04. Barrenlands to Croplands	05. Barrenlands to shrublands	06. No change: Other LC
Changes 2002 to 2013	WS1: Aba Gerima	278	94	11	19	0	495
	WS2: Zigba	322	110	10	63	0	451
	WS3: Yedemo	576	80	10	100	0	446
	WS4: Neber Wenz	204	86	25	5	0	483
	WS5: Laguna	596	222	48	23	0	1187
	Total area	1976	592	104	210	0	3061
Changes 2002 to 2019	WS1: Aba Gerima	273	170	113	20	75	246
	WS2: Zigba	317	182	94	64	72	228
	WS3: Yedemo/Robit	575	113	72	100	100	251
	WS4: Neber Wenz	198	167	136	6	74	222
	WS5: Laguna	578	488	321	28	154	506
	Total area	1940	1120	735	219	474	1454
Changes 2013 to 2019	WS1: Aba Gerima	388	84	102	1	75	246
	WS2: Zigba	494	80	83	1	72	228
	WS3: Yedemo/Robit	763	35	62	1	100	251
	WS4: Neber Wenz	304	90	111	1	74	222
	WS5: Laguna	831	306	273	6	154	506
	Total area	2780	594	631	10	474	1454

Table 7 Changes in the leaf area index (LAI) in five watersheds during the watershed intervention period (2002–2019)

Year	Watershed name	LAI (ha)				
		< 0.25	0.25–0.99	1.0–1.99	2.0–2.99	3.0–5.0
Year 2019	WS1: Aba Gerima	1	289	464	129	10
	WS2: Zigba	3	347	532	69	1
	WS3: Yedemo	1	413	653	133	6
	WS4: Neber Wenz	0	202	455	131	7
	WS5: Laguna	0	696	1202	167	5
	Total area	5	1947	3307	629	29
Year 2013	WS1: Aba Gerima	14	358	503	17	1
	WS2: Zigba	20	592	331	9	0
	WS3: Yedemo	29	720	438	19	0
	WS4: Neber Wenz	2	429	346	17	1
	WS5: Laguna	16	1307	717	28	1
	Total area	81	3407	2335	91	3
Year 2002	WS1: Aba Gerima	67	735	91	0	0
	WS2: Zigba	147	753	51	0	0
	WS3: Yedemo	196	934	77	0	0
	WS4: Neber Wenz	15	677	102	0	0
	WS5: Laguna	83	1763	223	0	0
	Total area	508	4862	546	1	1

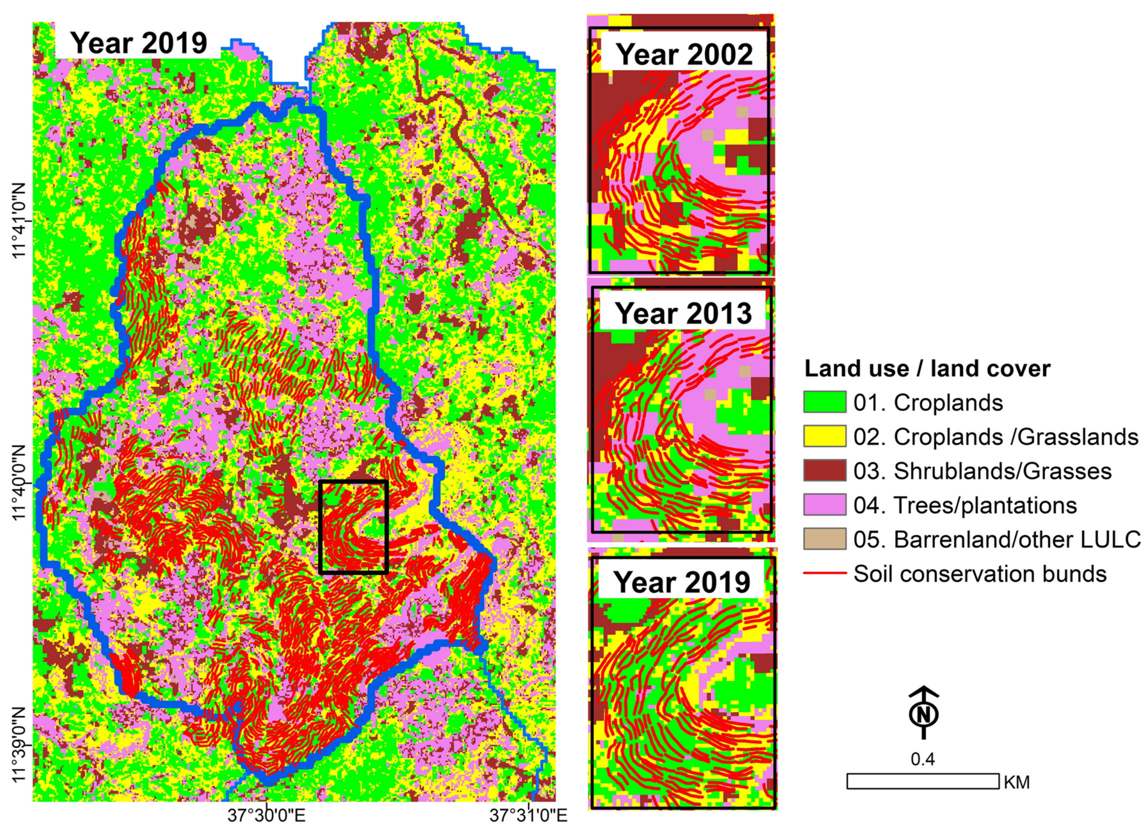

Fig.7 Soil conservation bunds (left) and contour-wise changes in LULC (right) for 2002, 2013 and 2019

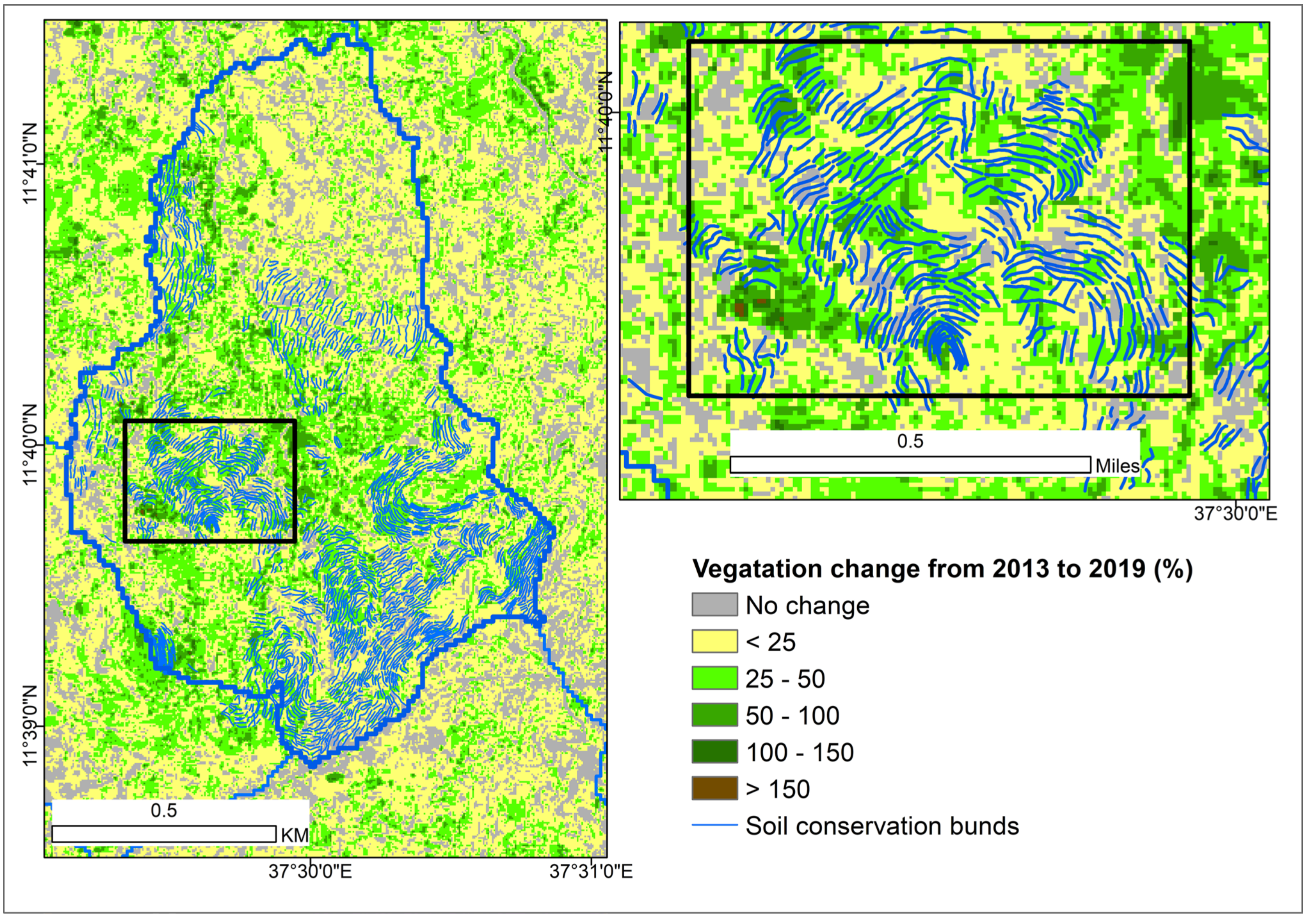

Fig. 8 Vegetation cover and percentage change in vegetation cover from the start of the interventions in 2013 (pre-intervention) and 2019 (post-intervention)

were converted into cropland and 474 ha of barren lands into shrublands i.e. vegetated. Moreover, large areas of crop lands (2780 ha) were maintained without any conversion in the period 2013–2019. It can be observed that nearly 1454 ha of other land-cover types sustained no change.

The changes in contour-wise LULC can be observed in Fig. 5. These can be attributed to an increase in the density of soil conservation bunds. The contour data collected during 2019 were overlaid on LULC maps, and changes were noted. There were significant changes from other classes to the crop class. For the years 2002 and 2013, most of the area was covered with trees/plantations and shrub land/grasses, but after the interventions happened during 2012–2018, the change from other LULC to croplands can be observed in the 2019 map.

Change in vegetation cover

The impacts of watershed interventions were assessed using the NDVI vegetation cover indicator to gauge the effectiveness of rehabilitation. This demonstrated a positive change (50–150%) in vegetation cover (Fig. 8) relative to the starting period of the interventions. A high rate of vegetation cover was observed in gully rehabilitation and area exclosures where intensity of land use was managed by disallowing no livestock grazing and adopting cut and carry grazing system. Fodder shrubs planted on soil conservation bunds also contributed to the improvement in vegetation cover. The effectiveness and level of impact of restoration practices were also assessed by comparing the extent of vegetation degradation before and after intervention. This implies that targeted implementation of NRM practices can lead to improved and productive land uses by adopting appropriate land management and regulated use, by shifting from free grazing to no grazing.

Spatial distribution of LAI changes in model and neighbouring watersheds

The leaf area index varied across the study area from 0 to 5.0 (Table 7; Fig. 9). The LAI for different years was spatially mapped for the five watersheds. The spatial distribution

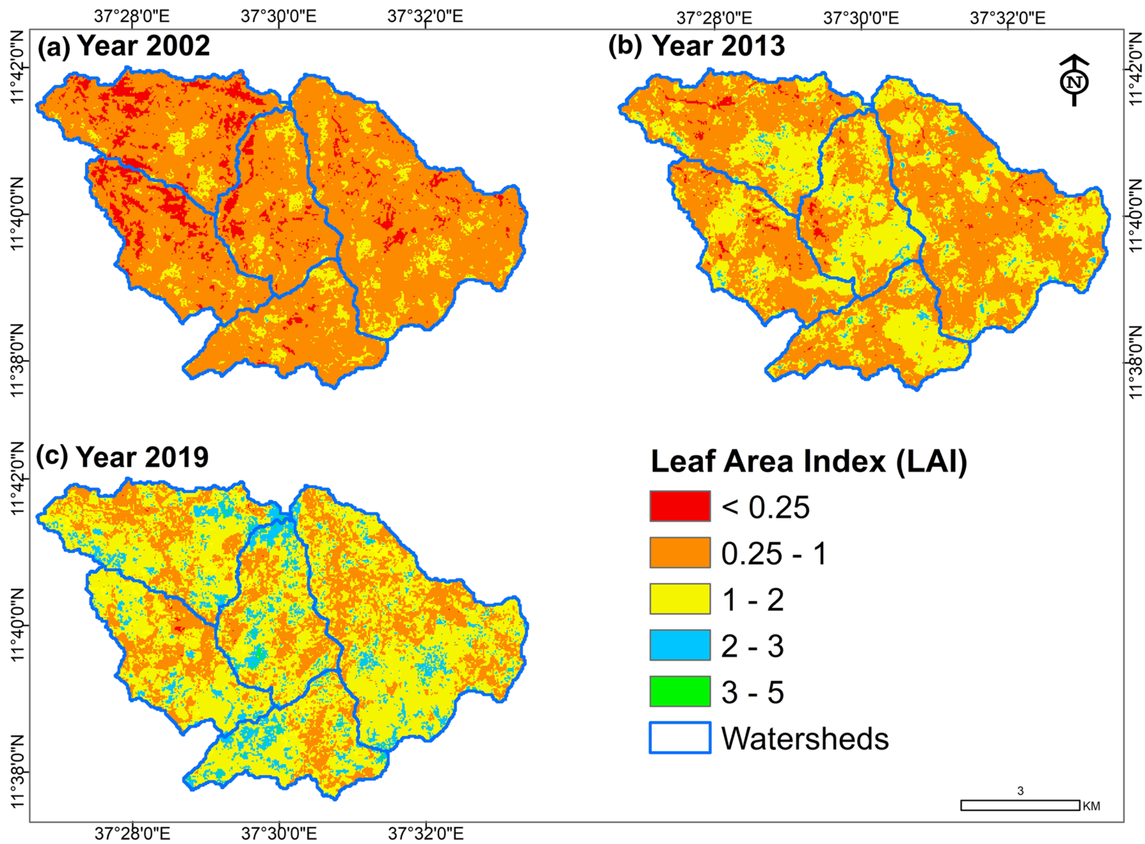

Fig. 9 Leaf area index (LAI) in the study watersheds in **a** 2002 (pre-interventions); **b** 2013 and **c** 2019 (post-interventions)

of the changes is shown in Fig. 10. The LAI characterises the plant canopy. We can observe major changes in the 1–1.99 and 2–2.99 LAI value bands in which crop land lies. With the increase in cropping area, the LAI in these areas increased too. Overall during 2002–2019, there was a 546 ha to 3307 ha in 1–1.99 LAI range. In 2–2.99 LAI range, it increased from 1 to 629 ha, which is a highly positive change. In <0.25 and 0.25–0.99, LAI ranges the cropping area decreased because the grasslands and shrublands were changed to crop lands. Overall, there was a vegetation cover shift from low LAI values to higher LAI values over the 15-year period (2002–2019). For LAI <0.25 range, the decrease was from 8.5% to <1%, and for the 0.25–0.99 LAI range, the decrease was from 82 to 33%. However, there was an increase in the vegetation area from 9 to 56%, 0 to 10.6% and 0 to 0.5% for the LAI bands 1–1.99, 2–2.99 and 3–5.0, respectively (Table 7).

In order to assess the impact of watershed interventions on in vegetation cover in the model or treated watershed and the four untreated neighbouring watersheds, a separate analysis was conducted for each watershed (Fig. 9). We found that the rate of decrease in LAI values <0.25 from the year 2002 to 2019 was 7.4% at WS1 (model watershed), 2–4% for upstream neighbouring watersheds (WS4 and WS5) and 15% for downstream watersheds (WS2 and WS3). Similarly, area coverage by LAI values 0.25–0.99 decreased by 43% for WS2 and WS3 and 50–59% for WS4 and WS5. On the other hand, the rate of increase for LAI values between 1 and 1.99 was about 42% in the model watershed (WS1) and 48–50% in the neighbouring watersheds. This implies that the model watershed has a role to play in slowing down and regulating the alarming land-use changes in the other watersheds. More importantly, the model watershed (WS1) exhibited a more significant change in the higher LAI range (2–2.99 and 3–5.0) than the adjacent watersheds. In the model watershed, the rate of increase was 15% and 1.12% for LAI bands, 2–2.99 and 3–5.0, respectively, while it was 7–11% and 0.1–0.8% respectively for the neighbouring watersheds. When comparing watersheds before (2013–2002) and after the (2013–2019) watershed intervention, the past trend in LULC change (i.e. LAI < 2) continued for the untreated neighbouring watersheds while there was a reverse trend in the model watershed. For LAI > 2, there was an increasing trend, but a higher rate for the model watershed resulted in

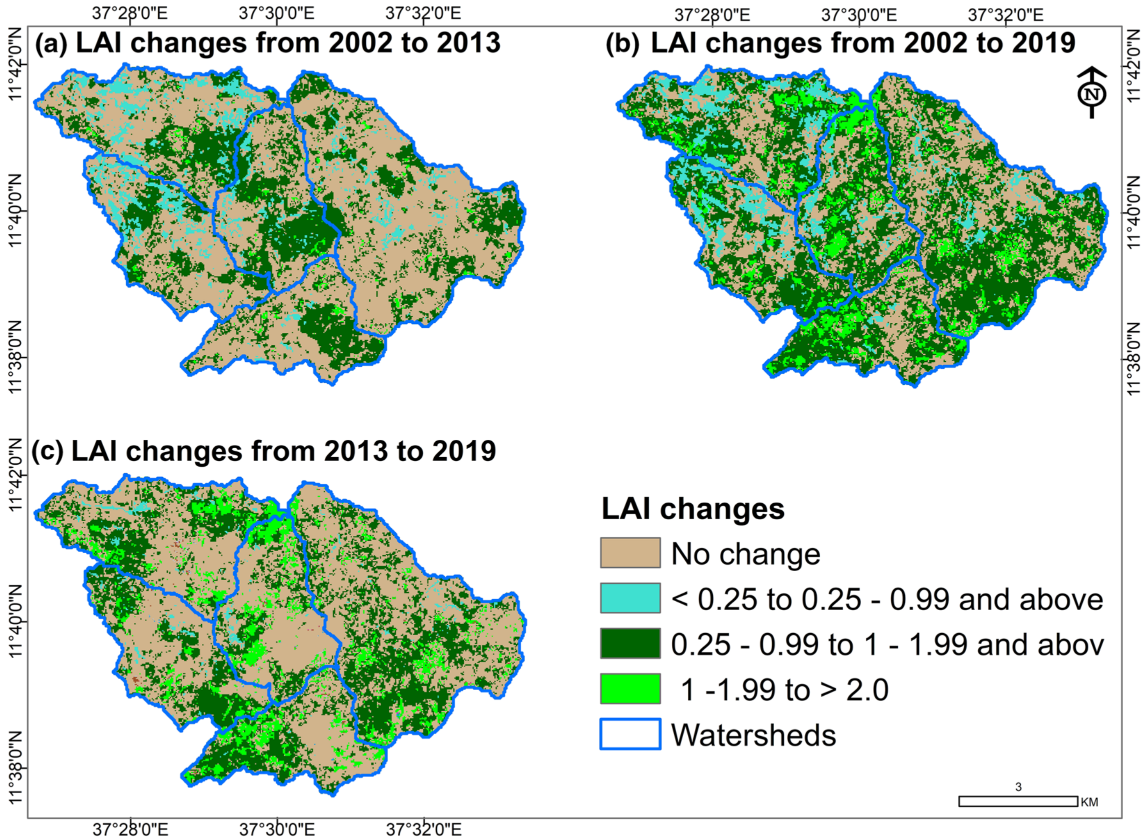

Fig. 10 Spatial distribution of LAI changes in the study watersheds from **a** 2002 to 2013; **b** 2002 to 2019 and **c** 2013 to 2019

improved vegetation cover and expansion of more intensified home garden practices with fruit production.

Conclusion

The impacts of watershed interventions over time can be assessed by combining field monitoring and remote sensing approaches. In the study watershed, implementation of integrated practices resulted in improved on-site soil and vegetation systems. In particular, multiple NRM interventions resulted in substantial change in vegetation cover which in turn reduced the rate of land degradation from 8% to 4.6%. Although this study was based on limited data other than LULC to demonstrate the overall impact and to compare with non-treated neighbouring watersheds, the LULC change analysis between the model watershed and four neighbouring watersheds showed that integrated watershed interventions reverse or regulate the trend of land-use changes from perennial vegetation to cropland (LAI < 2) and at the same time increase cropland intensification by increasing access to water. The neighbouring watersheds indicated a trend of conversion of perennial vegetation to crop land, whereas in the model watershed itself, there was a decreasing rate of conversion accompanied by an increasing rate of degraded land restoration. The study also illustrated the contour level LULC changes during the intervention years, distribution of sediment retention of soil conservation bunds and a significant percent change in vegetation cover. Despite these positive impacts of watershed interventions, this study has some limitations regarding extensive field measurement data to potentially demonstrate the linkages of impacts of treated watersheds and the environmental and socio-economic implications of adjacent untreated watersheds. Thus, further research should investigate the effects of interventions beyond the target watersheds and in adjacent watersheds by considering more environmental and economic data and other ecosystem service indicators with the help of extensive ground data and high-resolution satellite imagery.

Appendix 1: Accuracy assessment of LULC areas for the years (a) 2019; (b) 2013 and (c) 2002.

Classified data	Ground data					Row total	Classified total	Number Correct	Producer accuracy (%)Producer accuracy (%)	User accuracy (%)	Kappa
	CL_1	CL_2	CL_3	CL_4	CL_5						
(a) 2019											
CL_1	51	1	0	3	0	55	55	51	94.44	92.73	0.87
CL_2	2	15	4	0	0	21	21	15	83.33	71.43	0.66
CL_3	0	1	13	0	1	15	15	13	74.47	86.67	0.84
CL_4	1	1	0	22	1	25	25	22	84.62	88	0.85
CL_5	0	0	0	1	1	2	2	1	33.33	50	0.49
Reference totals	54	18	17	26	3	118	118	102			
Overall accuracy: 86.44%											
(b) 2013											
CL_1	58	2	0	1	0	61	61	58	92.06	95.08	0.90
CL_2	2	13	0	0	0	15	15	13	76.47	86.67	0.85
CL_3	0	2	8	0	1	11	11	8	88.89	72.73	0.71
CL_4	3	0	1	20	0	24	24	20	86.96	83.33	0.80
CL_5	0	0	0	2	11	13	13	11	91.67	84.62	0.83
Reference totals	63	17	9	23	12	124	124	110			
Overall accuracy: 88.71%											
(c) 2002											
CL_1	31	0	1	0	0	32	32	31	67.39	96.88	0.95
CL_2	0	7	1	0	0	8	8	7	70	87.5	0.86
CL_3	12	1	12	0	0	25	25	12	80	48	0.41
CL_4	1	1	0	32	0	34	34	32	100	94.12	0.92
CL_5	2	1	1	0	23	27	27	23	100	85.19	0.82
Reference totals	46	10	15	32	23	126	126	105			
Overall accuracy: 83.33%											

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