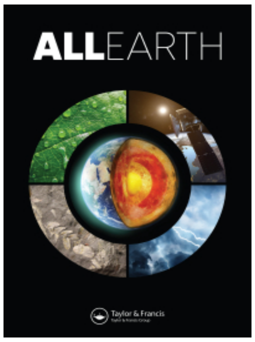

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RESEARCH ARTICLE

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Evaluating the impacts of watershed rehabilitation and irrigation interventions on vegetation greenness and soil erosion using remote sensing and biophysical modelling in Feresmay watershed in Ethiopia

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ABSTRACT
Soil erosion and subsequent land degradation undermine efforts to ensure food security and environmental sustainability in Ethiopia. The government of Ethiopia has implemented extensive soil and water conservation (SWC) programs in severely degraded and food-insecure areas of the country, in some cases integrated with subsequent or parallel irrigation development. However, the effectiveness of these interventions has not been extensively evaluated. This study, therefore, evaluates the performance and impacts of SWC practices in terms of improving vegetation greenness and reducing soil erosion in Feresmay watershed in Ethiopia. Long-term Landsat-based Normalised Difference Vegetation Index (NDVI), Revised Universal Soil Loss Equation (RUSLE), and Soil and Water Assessment Tool (SWAT) were used for change-detection analysis before and after the implementation of various SWC interventions. The results revealed the positive impacts of SWC interventions in improving the vegetation greenness and soil erosion reduction although the outcome varied by intervention. Increased vegetation greenness was observed largely in areas where area closure with catchment treatment (ACCT) and impacts of irrigation (IRR) interventions are dominant, while relatively little impact was observed at the watershed level analysis. Although these interventions helped to reduce soil loss, the results highlighted the need for more SWC interventions to minimise further soil loss.

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KEYWORDS
Feresmay; soil loss; evaluation; vegetation greenness; soil water conservation

1. Introduction

The Ethiopian economy primarily depends on agriculture, which contributed 35.5% of gross domestic product (GDP) in 2020 and employed 67% of the labour force in 2019 (World Bank, 2022; ILOSTAT 2021). Despite its economic contributions, the agricultural sector is vulnerable to climatic variability that is largely due to shifts in the timing of weather patterns and uneven rainfall distribution. According to Araya and Stroosnijder (2011), dry spells, short growing seasons, and lack of rainfall account for 80% of crop yield reductions and 50% of crop failures in parts of the country. Climate variability also contributes to high-intensity rainfall events that often instigate low infiltration and high surface runoff. Such events exacerbate soil erosion and land degradation, which in turn undermine agricultural productivity. The erosive power of rainfall is substantial, particularly in the northern highlands (Nyssen et al., 2005) and in the Central Rift Valley (Meshesha et al., 2012), as compared to elsewhere in the country (Haregeweyn et al., 2015). The situation has been severe in the highland areas of the country, where there has been extensive cultivation on steep slopes, deforestation, and overgrazing (Bewket, 2007).

The Ethiopian highlands are home to 88% of the country's population and 95% of its cultivable land and biodiversity. They have been characterised by severe soil erosion and land degradation, with an intensified decline in soil fertility and nutrients. These trends have placed enormous pressure on agricultural productivity and economic growth (B. G. J. S. Sonneveld & Keyzer, 2003; Bewket, 2007; Gebremichael et al., 2005; H. Hurni, 1985). Several studies have estimated the annual soil loss from sheet and rill soil erosion at various spatial scales (local to national) using plot experiments (Nyssen, Clymans, et al., 2009; Nyssen, Poesen, et al., 2009; Taye et al., 2013), the universal soil loss equation (USLE) (B. G. Sonneveld et al., 2011; Haregeweyn et al., 2012), process-based modelling (Betrie et al., 2011), and rill mapping and empirical estimation (Bewket & Sterk, 2003). Haregeweyn et al. (2015) conducted a literature survey of 25 case studies and estimated the mean annual soil loss from rill and sheet erosion to be 29.9 tons per hectare per year (tons $ha^{-1}yr^{-1}$) with a standard deviation of 30.2 tons per hectare per year

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(tons $ha^{-1}yr^{-1}$). Several other studies have estimated the rate of gully erosion, which varies with the stage and management conditions of a given watershed. According to these estimations, the mean annual rate ranges from 2.2 to 530 tons $ha^{-1}yr^{-1}$ in treated and untreated study areas, respectively (Adimassu et al., 2014; Daba et al., 2003; Frankl et al., 2013; Nyssen et al., 2008; Tebebu et al., 2010). Eroded soils are deposited in soil conservation and other hydraulic structures; this adversely affects the efficiency and functionality of the investments (Ministry of Agriculture and Rural Development MoARD and World Bank, 2007). These studies suggest that soil erosion is a serious issue in need of focused attention. Its severity has triggered the implementation of SWC structures to reduce soil erosion and improve agricultural productivity and food security in Ethiopia. Institutionalised SWC activities were initiated in the 1970s, and in fact, indigenous SWC practices date back to 400 BC (Haregeweyn et al., 2015).

More recently, concerted SWC activities have formed part of a public safety net program, in particular the Productive Safety Net Program (PSNP) that mobilises food-insecure populations in poor districts through 'food for work' activities (Tamene et al., 2006). Such programs are fully or partly supported by international donors such as the United States Agency for International Development (USAID) and the World Bank (Bezu & Holden, 2008). In addition to watershed rehabilitation, more recent investments include irrigation to increase household incomes and biodiversity protection through enhancing vegetation greenness in dry and wet seasons and improving groundwater recharge. The efficiency and sustainability of these interventions, however, depend on several factors including appropriate site selection during the implementation phase; proper maintenance; and public awareness creation concerning the benefits of the interventions to the community and the ecosystem at large (Ali et al., 2020; Bewket, 2007).

Several studies have quantified soil loss and land degradation at the watershed to basin scale in Ethiopia (Bewket, 2007; Haregeweyn et al., 2015; Hengsdijk et al., 2005; Nyssen et al., 2010). Others have measured the positive impacts of soil and water management practices on reduction in soil erosion (Gebremichael et al., 2005; Tefera & Sterk, 2010), land use change and associated soil losses (Bewket & Abebe, 2013; Mekuria et al., 2007; Tadesse et al., 2017), and changes in runoff (H. Hurni et al., 2005). Even though watershed conservation and irrigation interventions have been widely implemented in Ethiopian highlands for the past couple of decades to improve soil moisture and reduce soil erosion and runoff, few studies have evaluated the effectiveness and impacts of SWC interventions using remote sensing and process-based biophysical modelling approaches. Ali et al. (2020) used Landsat 7-based NDVI and measured streamflow and sediment load data to evaluate the benefits of SWC interventions in the Tana Beles watershed in Ethiopia. The study suggested that SWC activities positively impacted vegetation greenness in the watershed, although the improvement appeared to vary by season. The study further reported that the effectiveness of the interventions was more evident for degraded land than cultivated land (Ali et al., 2020). Considering the buffer area around the watershed to evaluate the benefits of the SWC interventions might overlook the spillover effects of the interventions and can be considered a limitation of this study. Alemayehu et al. (2009) similarly showed the benefits of watershed rehabilitation interventions and their impacts on soil erosion, runoff reduction and groundwater availability in the upper Agula watershed in the Tigray region. Since the impacts of watershed interventions vary based on agroecology and landscape, more research following a before- and after-interventions approach would help to understand the role of SWC practices and thereby improve their effectiveness. This study, therefore, evaluated the impacts of watershed restoration interventions in improving vegetative greenness and annual soil loss, enhancing moisture availability and duration as a proxy in the Feresmay watershed in the Tigray region of Ethiopia. Watershed-restoration interventions implemented in the Feresemay watershed include, area closure, catchment treatment, terracing, gabion check dams, water harvesting ponds, check dams, mini dams, and water points such as spring and hand-dug wells. These interventions were financed by the Ethiopian government and international donors such as USAID and the World Bank.

Watershed rehabilitation activities were monitored using high-resolution satellite images and other hydroclimatic and biophysical observations designed to detect changes in vegetation greenness, land use, and soil moisture and to measure changes in surface flow that result from enhanced subsurface storage after interventions. Recent advances in remote sensing, access to high-resolution satellite images, and cloud computing capabilities such as the GEE enabled more efficient evaluation at finer spatial resolution and for an extended time period (Ali et al., 2020). The study also used the RUSLE and SWAT to estimate annual soil loss in the watershed before and after the interventions and to evaluate the change in soil loss due to the interventions.

2. Material and methods

2.1. Description of the study area

The Feresmay watershed is located in Ahferom woreda in the northern Tigray region in Ethiopia. The watershed is bounded by the geographic coordinate

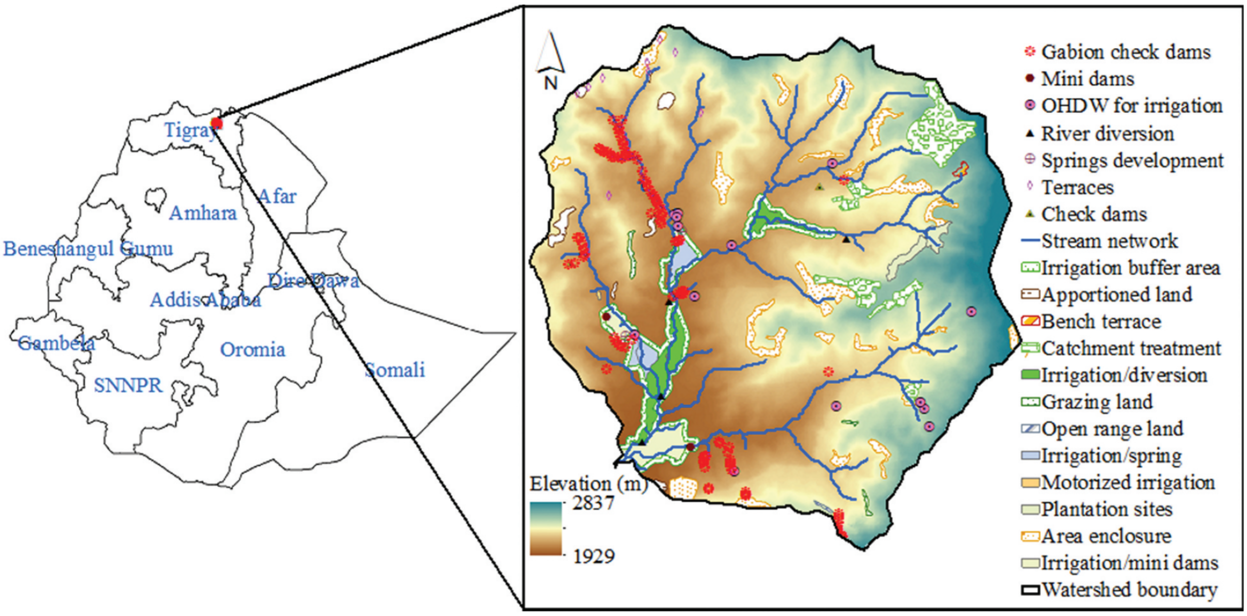

Figure 1. Location of the Feresmay watershed in the Tigray region in Ethiopia. The figure in the right shows the different SWC interventions implemented in the watershed.

of 14°12′07″– 14°14′52″ North and 39°06′14″ – 39° 08′26″ East (Figure 1). The total area of the watershed is ~77 km^2, and the altitude ranges from 1,929 m to 2,837 m. A large part of the watershed (>60%) is covered by slopes ranging from moderate to very steep (Table 1). The watershed includes three agro-climatic zones traditionally classified as Woina Dega (sub-humid), Dega (humid) and Wurch (very humid). The mean annual rainfall and temperature (1983 – 2016) were 650 mm and 17°C, respectively. The watershed has a bimodal rainfall pattern in which the main rainy season (locally called Kiremt) occurs from June to September and the small rainy season (locally called Belg) occurs between February to April. The agricultural practice in the watershed is primarily rainfed agriculture, and its productivity significantly depends on the seasonal rainfall variability.

Watershed degradation reduces agricultural productivity, thereby threatening the livelihoods of smallholder farmers and exacerbating food insecurity and malnutrition (Haregeweyn et al., 2012). Since Feresemay watershed is one of the most severely degraded watersheds in Tigray, the Relief Society of Tigray (REST) with support from the USAID initiated watershed rehabilitation and subsequent irrigation intervention investments in the watershed since 2001. The intensified watershed-management intervention began in 2008. The objective of implementing the watershed rehabilitation and irrigation interventions was to restore degraded lands, minimise the risks of disasters such as flash floods and droughts, improve food security, and enhance community resilience to climate shocks.

Figure 1 presents the different types of watershed conservation and irrigation practices implemented in the watershed. Depending on the extent to which the appropriate interventions are being implemented, we hypothesise that these interventions may positively impact and restore the ecosystem by reducing soil erosion and subsequent land degradation and improving water infiltration and groundwater recharge, thereby enhancing vegetation cover, reducing flood risk, and promoting afforestation. The irrigation intervention may, additionally, foster agricultural production during the dry season and improve food security and household income.

Table 1. Landscape classes based on slope, and the corresponding fractional area in Feresmay watershed.

No	Slope (degrees)	Fraction area (%)	Landscape type
1	0–3	18	Flat or almost flat
2	4–9	22	Gentle slopes
3	10–15	24	Moderate slopes
4	16–30	18	Steep slope
5	31–60	12	Extremely steep slopes
6	>60	6	Excessively steep slopes

2.2. *Data acquisition, and RUSLE factors computation*

This study used different biophysical data to assess changes in vegetation greenness and to estimate soil loss due to the SWC interventions in the study watershed. Detailed descriptions of the data are illustrated in the following subsections.

2.2.1. *Normalised Difference Vegetation Index (NDVI)*

NDVI time-series data were generated using 30-metre (m) spatial resolution and radiometric and atmospheric-corrected Landsat data at the top-ofatmosphere (TOA) reflectance. Red and near-infrared spectral bands were used to derive the NDVI time series from 1984 to 2020. NDVI data were used to monitor vegetation greenness. The Landsat scenes were acquired from the US Geological Survey (USGS) site through a Google Earth Engine (GEE) platform.

2.2.2. Rainfall and rainfall erosivity (R) factor

Rainfall data from the Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS) were used in this study due to the lack of observed meteorological data in the watershed. CHIRPS blends ground-based rainfall measurements with satellite observations to enhance better accuracy, particularly in the ungauged catchments and in areas where there are sparse and unevenly distributed rainfall gauging stations. Numerous satellite rainfall evaluation studies have shown that CHIRPS rainfall product performed better than others in certain watersheds in Ethiopia (Ayehu et al., 2018; Bayissa et al., 2017; Dinku et al., 2018; Duan et al., 2019; Funk et al., 2016; Gebremicael et al., 2019; Musie et al., 2019). For instance, Bayissa et al. (2017) evaluated five high-resolution rainfall products in the Upper Blue Nile Basin and reported the best performance of CHIRPS in representing the observed rainfall. Gebremicael et al. (2019) also reported the best performance of CHIRPS over the rugged topography of the Tekeze-Atbara basin where the current study watershed is located. Other studies (e.g. Ayehu et al., 2018; Dinku et al., 2018; Duan et al., 2019; Funk et al., 2016; Musie et al., 2019) have also supported the application of the CHIRPS rainfall product in hydrological and water management planning in Ethiopia. Accordingly, the time series of CHIRPS rainfall data were processed from 1984 to 2020 at 5 km spatial resolution and used as an input for RUSLE and SWAT models.

The rainfall erosivity factor (*R*) in the RUSLE represents the kinetic energy the rainfall possesses to detach and transport soil particles from one place to another (Simms et al., 2003). Accurate estimation of the *R*-factor depends on the availability of rainfall intensity measurement, which is not commonly accessible in many watersheds including the Feresmay watershed. In such situations, empirical equations were derived to compute the *R*-factor using average annual rainfall. For example, H. Hurni (1985) developed equation (1) to estimate the R-factor. This empirical equation is widely used to estimate the rainfall erosivity factor in Ethiopia and used to estimate R-factor (Figure 2a) in this study.

$$R = 0.562\mathrm{P} - 8.12 \tag{1}$$

Where R represents rainfall erosivity factor ($\mathrm{MJ\,mm\,ha^{-1}\,h^{-1}\,yr^{-1}}$), and P is the annual rainfall in mm.

2.2.3. Soil and soil erodibility factor (K-factor)

Africa Soil Information Service (AfSIS) soil data of 250 m spatial resolution was used in this study. Soil properties such as soil texture, organic carbon, rock fragment, and other properties were obtained for up to six soil-depth layers and used for SWAT model setup and to compute soil erodibility *K*-factor. *K*-factor measures the sensitivity of soil particles to be detached and transported by rainfall and runoff. Direct measurement of soil erodibility factor is costly and requires more time. Wischmeier and Smith (1978) proposed an empirical (Equation 2) method to quantify the *K*-factor based on soil properties and used in this study. Equation 2 applies when the silt and very fine sand content add up to less than 70% of the soil particle size distribution. The total silt and very fine sand content in the study watershed range between 59 and 65.

$$K = \frac{\begin{array}{c}[0.00021*M^{1.14}*(12-OM)+\\ 3.25*(s-2)+2.5*(p-3)]\\ *0.1317*e^{-0.04*(R_c-10)}\end{array}}{100} \tag{2}$$

where K is soil erodibility ($\mathrm{t\ h\ MJ^{-1}mm^{-1}}$), OM is the percent organic matter (%), M is particle-size parameter, s is the soil structure class with a default value of 2; and p is the permeability class ranges with a default value of 3; Rc is surface rock fragment cover.

M is calculated using, $M = (m_{silt} + m_{vfs}) * (100 - m_{clay})$
where $m_{silt}, m_{vfs}, and m_{clay}$ are percent silt, very fine sand and clay contents, respectively.

The percent organic matter content, OM, is calculated:

OM = 1.72*orgC, where orgC is the percent organic carbon content of the layer (%). The map of K-factor is shown in Figure 2b.

2.2.4. Land use and Cover management factor (C-factor)

The land-use maps were prepared based on 30 m spatial resolution Landsat images taken on 11 October 2006 and 11 October 2019. Severe to extreme drought events were not recorded in these years in the study watershed and Ethiopia in general. The annual rainfall in these years was higher than the long-term average rainfall. Supervised image classification was applied to classify the land use classes in the watershed. High-resolution satellite images in the GEE platform were used to create training samples to classify the land use types. The classification resulted in seven major land use classes in both years (Figure 2c).

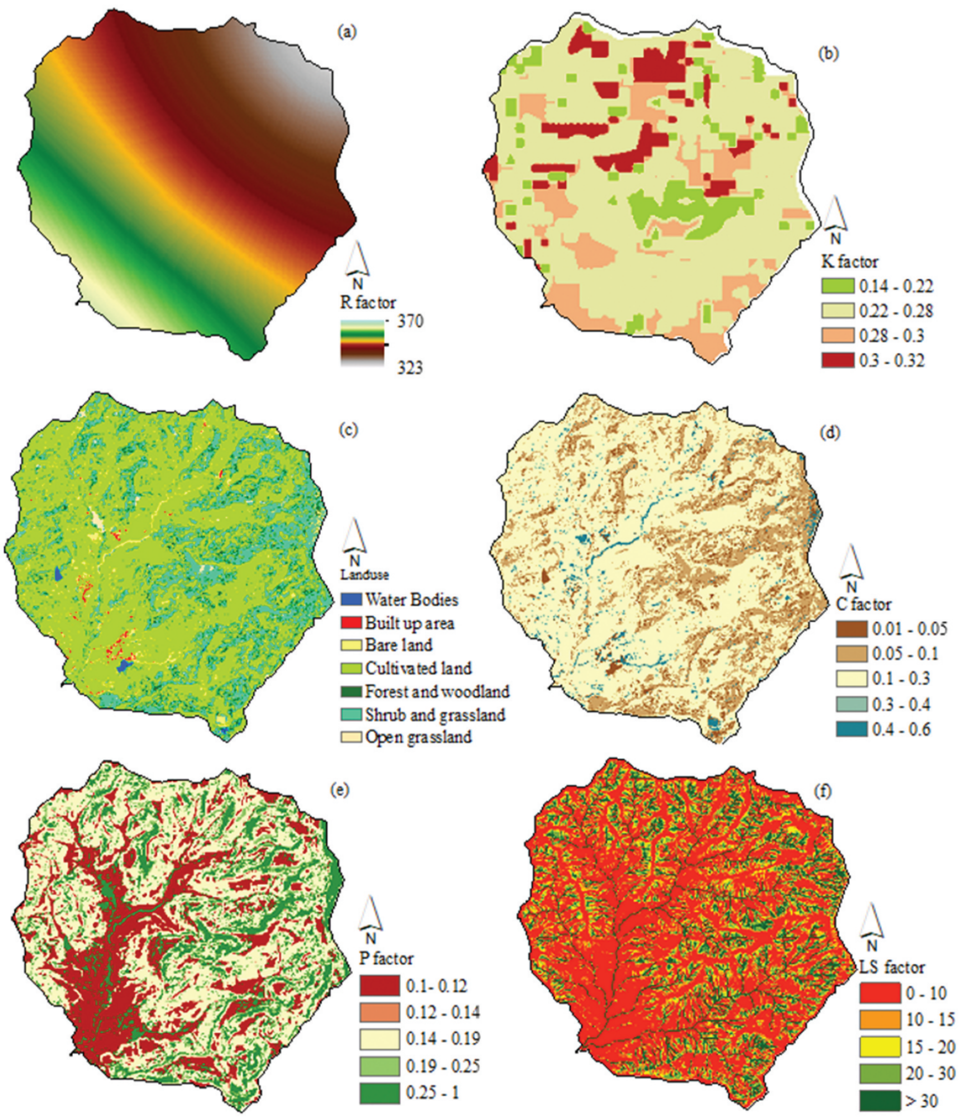

Figure 2. Spatial distribution and estimation of R-factor (a), K-factor (b), land use (c), C-factor (d), P-factor and LS-factor (f) across the watershed.

Agricultural land was the dominant land use type followed by shrubland.

The C-factor informs the effect of cropping and management practices on erosion rates due to raindrops and overland flow (Simms et al., 2003). It basically measures the proportion of soil loss from each land use cover to the soil loss from a continuously tilled bare fallow land (Simms et al., 2003). The C-factor values for different land use types in the study watershed were adapted from H. Hurni (1985), Wischmeier and Smith (1978), and other sources (Table 2).

Table 2. Land use (LU) and the corresponding cover management factor (C-factor) values.

LU/LC	C-factor	Source
Cultivated Land	0.2	H. Hurni (1985)
Forest and woodland	0.02	H. Hurni (1985)
Shrub and grassland	0.10	Wischmeier and Smith (1978)
Open grassland	0.11	Eweg and van Lammeren (1996)
Built up area	0.01	H. Hurni (1985)
Bare soil	0.6	BCEOM (1998)
Water bodies	0.01	BCEOM (1998)

2.2.5. Management support practice factor (P-factor)

The *P*-factor measures the reduction in soil loss attributable to land management and soil conservation practices (K. Hurni et al., 2015). Prior knowledge of the spatial distribution of the existing physical soil conservation structures is required to calculate the *P*-factor. Even though the location of each intervention in the study watershed was known, detailed information on the length, width and spacing between the interventions was not available. Therefore, slope and land use information were used to estimate P-factors for the study watershed (Table 4). Land uses were clustered into agricultural and nonagricultural broad classes based on Wischmeier and Smith (1978) recommendations (Table 3). The agricultural land use types

Table 3. P-factor values determined based on land use and slope combinations.

Land use type	Slope (%)	P-factor
Agriculture/cultivated land	0–5	0.10
	5–10	0.12
	10–20	0.14
	20–30	0.19
	30–50	0.25
	50–100	0.33
Other land	All	1.00

Table 4. Percentage change in vegetation greenness before and after the interventions, for the aggregated groups of interventions in Feresmay watershed.

Type of treatment	Percentage change before and after intervention
ACCT	20%
IRR	28%
SWC	16%
Watershed level	13%

were further classified into six classes based on the slope (Table 3). *P*-factor values were determined for each subcategory (Table 3) in accordance with the recommendations of Wischmeier and Smith (1978). Figure 2e shows the area and percent distribution of the P-factor values across the watershed.

2.2.6. Digital elevation model (DEM) and slope length-gradient (LS) factor

The impact of slope length and steepness on sheet and rill erosion is rendered by the LS factor in RUSLE (Simms et al., 2003). The LS factor was calculated using the upslope contributing factor (A) and slope angle (B) as set forth in equation 2. Topographic information such as A and B was extracted from a 30 m resolution DEM data set acquired from the Shuttle Radar Topography Mission (SRTM).

$$T = \left(\frac{A}{22.13}\right)^{0.4}\left(\frac{\sin B}{0.0896}\right)^{1.3} \tag{3}$$

Where A is the upslope contributing factor, which is a product of the flow accumulation and pixel size (30 m), and B is the slope angle.

2.3. Method

Basically, this study employed two approaches to assess the impacts of SWC interventions in the Feresmay watershed (Figure 3). One of the approaches evaluated the impacts of the interventions on vegetation greenness using remote sensing data, while the other approach evaluated the impacts of the

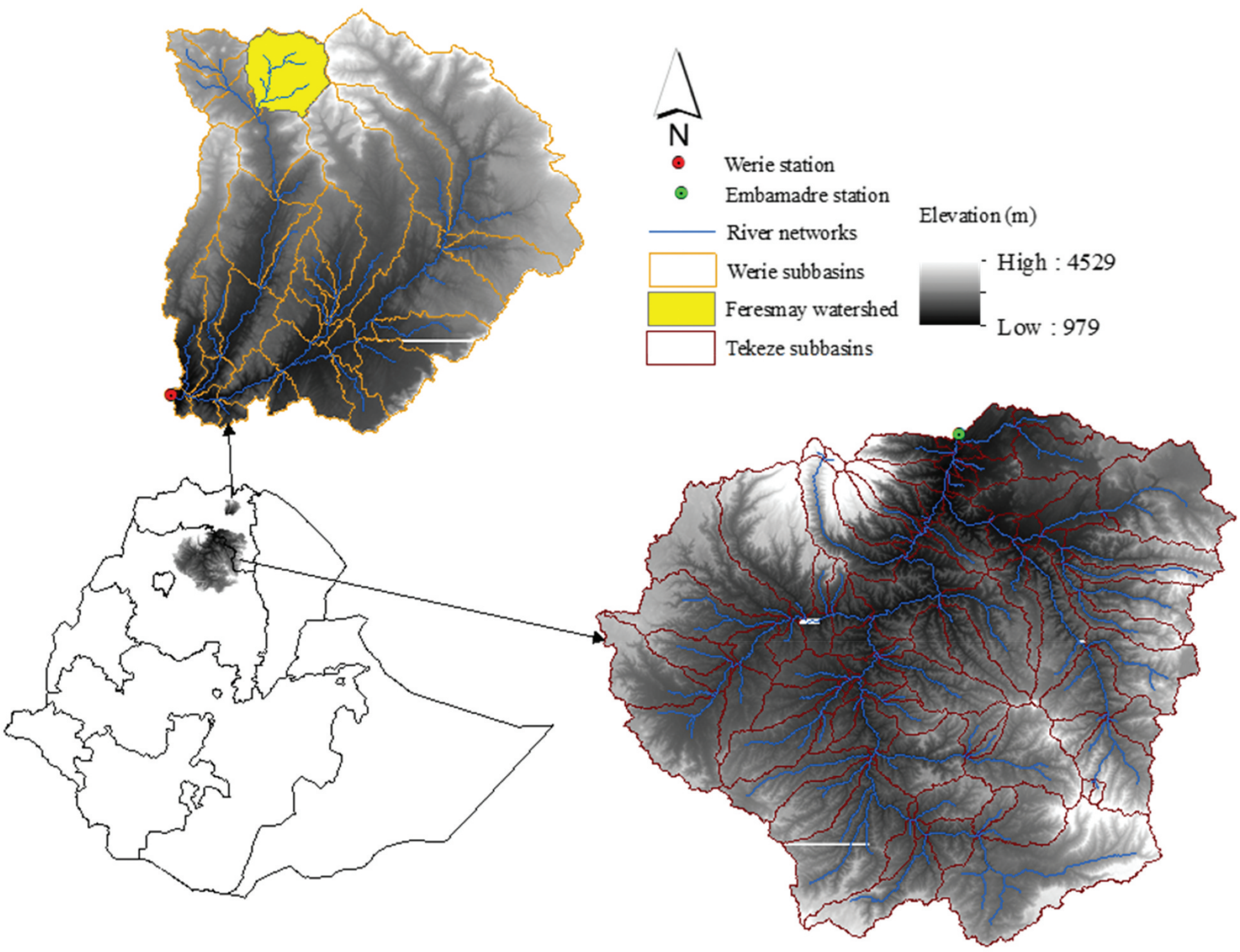

Figure 3. SWAT subbasin discretization for Tekeze basin and Werie watershed used for model calibration and cross-validation.

interventions on soil erosion using the RUSLE empirical equation and SWAT hydrological model. As previously indicated, *before*- and *after*-intervention periods were considered for the change detection analysis. Based on the information gathered from the implementer (REST), the threshold year was decided and used to truncate the data before- and after-intervention. In addition, the high-resolution satellite images in the Google Earth Explorer platform were used to verify the years of the implementation of some of the physical structures (e.g. mini dams). Although the implementation of the SWC interventions commenced in 2001, extensive interventions have been implemented since 2008. Consequently, 2008 is used as a - threshold year for the change detection analysis ('before' and 'after' interventions).

Besides, trends of monthly, seasonal, and annual rainfall were analysed before and after the intervention periods using CHIRPS rainfall data to explore any association between changes in vegetation greenness and soil erosion due to change in rainfall. The average areal rainfall was extracted using the boundaries of each watershed rehabilitation activity (i.e. ACCT, IRR and SWC), and watershed scale. The main rainfall season locally called 'kiremt' (June to September), which contributes more than 70% of the annual rainfall, was considered for the seasonal trend analysis. Several watersheds in Ethiopia receive a large amount of rainfall during the kiremt season and its failure often prompts drought conditions that cause socioeconomic damages (Viste et al., 2013). Thus, the trend analysis was carried out using linear regression and a non-parametric modified Mann – Kendall statistical test (Yue & Wang, 2004). This analysis helps to identify the impacts of rainfall on the vegetation and soil erosion. The Mann-Kendall statistical test at 95% ($\alpha = 0.05$) level of significance was used to test the monotonic upward or downward trends in rainfall extracted at the aggregated intervention sites.

2.3.1. Vegetation greenness change detection

Remotely sensed images before and after 2008 were used to assess the change in vegetation greenness due to the interventions. The enhancement in vegetation greenness was also evaluated during the dry (November to February) and wet (June to September) seasons and in selected historic drought years both before (1984, 1985, and 1991) and after (2009, 2013, and 2015) the interventions. These drought years were identified using standardised long-term annual rainfall anomalies for the watershed. Changes in vegetation greenness can be considered as a proxy for changes in soil moisture and soil fertility, which in turn impacts agricultural productivity and food security in the watershed. The change detection analysis was carried out at and in the vicinity of the intervention sites as well as at the watershed-level to evaluate the on-site and watershed-level impacts. REST prepared the development map, which shows the extent and locations of the interventions, by conducting a field survey of the study watershed. Although there are various watershed conservation interventions implemented in the watershed, this study largely focuses on evaluating the impacts of area closure combined with enrichment plantations; other SWC interventions such as hillside and bench terraces, soil and stone bunds, different in situ water harvesting micro-basins, and various gulley rehabilitation interventions promoting vegetation greenness; and irrigation from sources including mini-dams, diversion weirs, hand-dug wells and springs. Impact analysis was done based on the following aggregation levels of the interventions:

(1) Area closure with catchment treatment (ACCT) with enrichment plantation. Area closure often benefits to restore the overall ecosystem since it isolates the interference of humans and other actors contributing to land degradation.
(2) Irrigation plus farmland treatment (IRR). This includes all irrigation sites where the sources of water are mini dams, diversion weirs, hand-dug wells, springs, canal extensions, or night storage, and treated with SWC practices. These irrigation intervention infrastructures were developed to supplement rainfed agriculture, which is vulnerable to climate shocks (Gebregziabher et al., 2009); impacts of the irrigation interventions on the buffer area are included in the change detection analysis since the main purpose of irrigation application is to supplement water obviously enhancing vegetation greenness.
(3) Various physical soil and water conservation (SWC) practices, which include all types of bunds, terraces, check dams, and moisture-harvesting micro basins;
(4) Watershed-level analysis that includes all treated and untreated areas.

High spatial and temporal resolution Landsat level 2 geometrically corrected near-infrared (*NIR*) and red (*RED*) bands were used to compute the time-series values of 16-day NDVI as shown in equation 3. The images with cloud coverage of greater than 3% were filtered out from further use. Apparently, the NDVI time-series data were grouped into before (1984 to 2007) and after (2008 to 2020) the interventions and used for the change detection analysis.

$$NDVI = \frac{(NIR - RED)}{(NIR + RED)} \tag{4}$$

The areal average NDVI values were extracted for each treated subareas and percentage change in vegetation greenness was assessed using long-term average vegetation greenness in each period.

2.3.2. Annual soil loss estimation using RUSLE

This study also employed the RUSLE method to estimate the impacts of the SWC interventions on annual soil loss. The analysis was conducted for the periods before and after the interventions to assess changes in annual soil loss attributable to the interventions. The interventions often have the potential to modify the land use of a watershed, although the impacts vary depending on mainly the area coverage and type of interventions. For example, area closure of degraded land may restore the vegetation cover and thereby may improve the land cover, which intercepts the raindrop impact on the soil surface and hence reducing soil erosion. The widely used RUSLE empirical equation (equation 4) was applied to estimate the annual soil loss in 2006 and 2019 to represent the losses before and after intervention. Land use change is a gradual process, and these years may be considered representative of the change detection analysis. The RUSLE model estimates sheet and rill erosion losses; however, it has limitations in estimating soil deposition and losses due to gully erosion. Nonetheless, the model has been applied in several watersheds and provided reasonable soil loss estimations (Renard et al., 1991).

$$ASL = R.K.LS.C.P \tag{5}$$

where ASL is annual soil loss in (ton. $ha^{-1}year^{-1}$), R is rainfall erosivity factor, K is soil erodibility factor, LS is topographic factor, C is cropping, and land-cover factor and P is conservation support practice factor.

All the RUSLE parameters were assumed to be the same before and after the implementation of the interventions except the R-factor, C-factor, and P-factor, which vary with rainfall, land uses/land covers and conservation practices.

2.3.3. Annual soil loss estimation using SWAT

Feresmay is an ungauged watershed, and we used the model parameters transfer approach for reasonable model simulation of the water balance components in the watershed. The SWAT model was first calibrated (1996–2001) and validated (2002–2007) using monthly streamflow data at the Embamadre gauging station (Figure 3), which is near the Tekeze hydropower dam. The default model parameters were manually fine-tuned; auto-calibration was then carried out using the Sequential Uncertainty Fitting (SUFI-2) in SWAT-CUP (SWAT Calibration and Uncertainty Program). The model performance was evaluated using the Nash-Sutcliffe model efficiency coefficient (NSE), the coefficient of determination (R2) multiplied by the regression slope, b, and the percent bias (PBIAS). Then, the model parameters were transferred to the Werie watershed, which has similar agroclimatic conditions and 3 years of streamflow record (2003–2005) for cross-validation. The study watershed is located within Werie watershed. In addition to model calibration using streamflow, monthly sediment concentration data at Tekeze dam from 2002 to 2006 were used to calibrate sediment controlling parameters. The sediment concentration data were generated using the rating curve developed using measured sample data for 100 continuous days in 2005/2006 (Welde, 2016).

Eventually, a separate model setup and simulation were carried out in the Feresmay watershed using the transferred model parameters and further soft calibration was made based on soil loss estimation of previous studies in the basin. Then, the different SWC interventions were integrated into the model to assess their impact on soil loss after the interventions. The SWC interventions were introduced into the model by modifying the model parameters per previous studies' recommendations and expert judgement based on the authors' vast experience in SWAT modelling. SWAT model calibration and validation results and the summary of the different types of interventions and adjusted model parameters are presented in supplementary material. Figure 4 summarises an overview of the workflow and methods followed in this study.

3. Result and discussion

3.1. Vegetation greenness change-detection analysis

The non-parametric Mann – Kendall trend analysis of the rainfall indicated no significant upward trend in the monthly, seasonal, and annual rainfall at 95% ($\alpha = 0.05$) level of significance (Appendix A). Overall, the trend analysis illustrates unremarkable changes in rainfall after the interventions as compared to the period before the interventions. Therefore, change in rainfall is not a substantial reason for changes in vegetation greenness and other biophysical variables in the study watershed.

The Box and Whisker plots in Figure 4 show the summary of NDVI before and after the interventions in the areas treated with ACCT, IRR, and SWC, as well as at the watershed level. The long-term median values are shown by the median marker lines in each box and for each site. A visual comparison of the median values indicates relatively higher after implementation of the interventions than before. Table 4 summarises the percentage change in vegetation greenness after and before the interventions for each aggregated treatment. The mean values of NDVI were used to derive the percentage change in the average NDVI after and before the interventions. ACCT showed improved vegetation greenness with a percentage change of 20%, while relatively a higher percentage change value (28%) was observed in the irrigation buffer area (IRR). Although the vegetation greenness due to irrigation application is relatively higher, the change is not

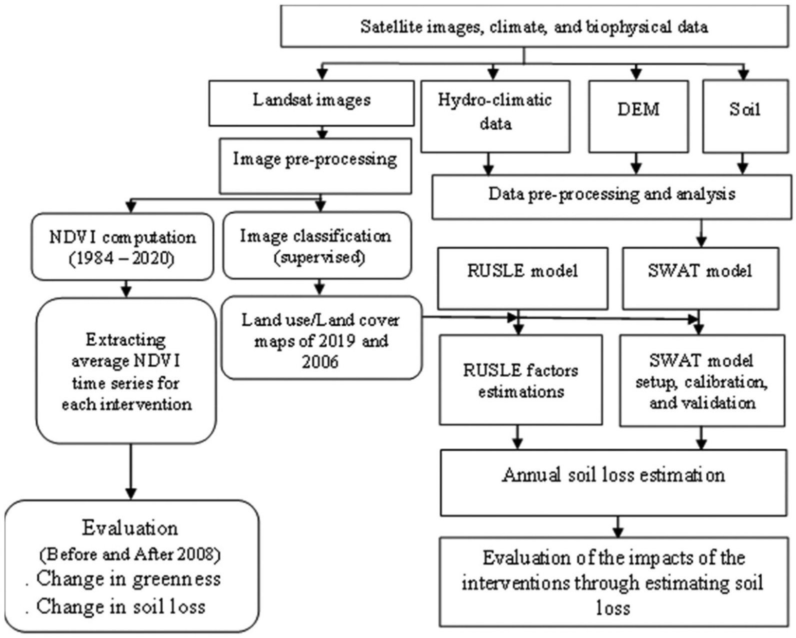

Figure 4. Workflow chart of the approaches followed in this study.

as large as expected since irrigation provides direct and presumably sufficient water to enhance greenness relatively more than the other interventions. There might be several interplaying factors including largely the reflectance of the vegetation/crops cover to the radiant energy (which varies per type of vegetation) and a lack of satellite observations during the plant growing stage where the leaf area index (LAI) would be highest. This should be validated with ground observations and field measurement data in future studies. Watershed-level analysis revealed an overall improvement in vegetation greenness across the watershed, as indicated by the improved percentage of greenness after the interventions. The change in greenness due to irrigation is used in this study as a reference; otherwise, irrigation application certainly improves greenness. Other SWC structures also benefited from enhancing vegetation greenness by 16%. The area of the treated part of the watershed considered in this analysis was about 16%. Thus, the watershed-level improvement in vegetation greenness may indicate a spillover or buffering impact in the untreated parts of the watershed. The community and farmers might have implemented other SWC structures, which might not be included in the development map. Given the assumption of all other factors – such as type of vegetation and its density, slope, management practices, extent of treatment, accuracy of the satellite observation, and other biophysical factors – as constant, the overall result indicated the benefits of watershed treatment in terms of restoring vegetation greenness and cover. The qualitative and quantitative assessments that use the household-level survey may augment the remote sensing-based assessment of the change-detection analysis of the greenness that follows from each intervention. Figure 4 also shows outlier values for each plot, indicated by black dots. The improvement in vegetation greenness may be considered as a proxy for increases in actual evapotranspiration (ET), soil moisture, and perhaps groundwater recharge due to the interventions, which may be verified using future ground observation measurements. The positive impacts of community-based watershed rehabilitation activities were reported by Siraw et al. (2020); significant improvement in vegetation greenness was identified in the study watershed as compared to the control watershed in which no interventions were implemented. Hishe et al. (2017) also reported positive impacts of area enclosure in terms of restoring land degradation and enhancing in vegetation greenness in the study region. The results reported in these studies support the findings of this study.

Seasonal average NDVI plots were generated for the wet (June to November) and dry (December to May) seasons for the treated subareas both before and after the interventions (Figure 5). The percentage change in vegetation greenness for both the wet and dry seasons is shown in Table 5. The analysis revealed an overall improvement in greenness in the treated subarea during both the dry and wet seasons, presumably due to the interventions; however, the watershed-level analysis showed an 18% reduction in greenness during the rainy season. This might be due to a relatively sufficient rainfall prior to the interventions to flourish vegetation greenness. In addition, the quality of the images and less number of observations during the wet season might also have contributed to the reduction in greenness at the watershed level during the wet season. Traditional irrigation practices before the intervention and perhaps varying years of implementation might

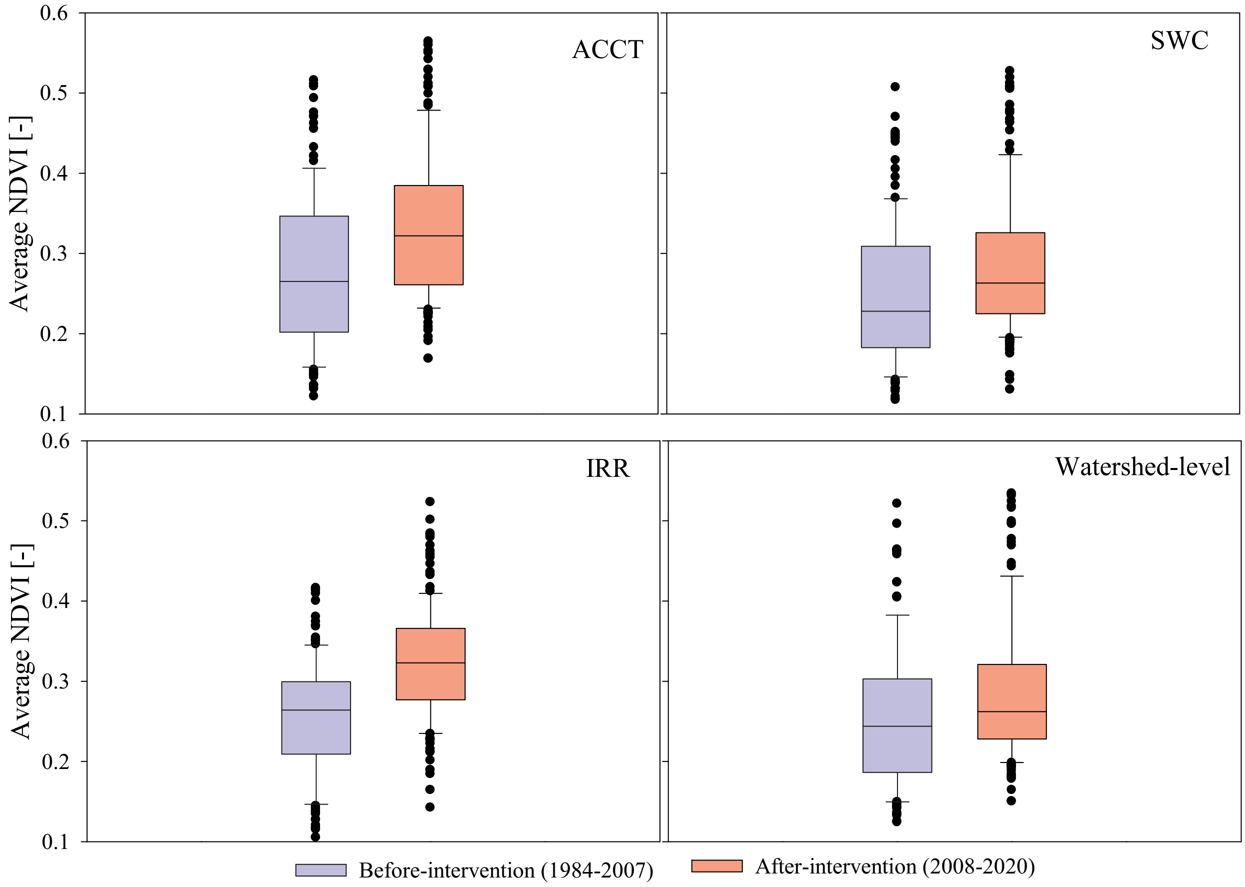

Figure 5. Box and Whisker plots in Feresmay watershed showing NDVI before and after the interventions in the treated area.

Table 5. Percentage change in vegetation greenness before and after the interventions during dry and wet seasons.

	Percentage change before and after intervention	
Type of treatment	Dry season	Wet season
ACCT	18	9
IRR	26	19
SWC	10	8
Watershed level	7	−18

contribute to enhancing the vegetation greenness at watershed level analysis.

As expected, the highest percentage of changes were observed in the irrigation buffer area during the dry and wet seasons (26% and 19%, respectively). At the same time, ACCT improved the greenness by 18% and 9% during dry and wet seasons, respectively. Vegetation greenness also improved during dry and wet seasons in the areas where SWC activities were implemented; however, these changes were less drastic relative to the other interventions. This is to be expected, given that the primary purpose of SWC practices is to control soil erosion. The improvement in greenness during the dry season indicates the positive impacts of the interventions in enhancing water storage capacity and soil fertility, and thus the residual soil moisture available during the dry season. The abundance of perennial and deep-rooted vegetation and excess residual soil moisture in area closure sites may also enhance vegetation greenness. Based on data collected from the implementer, indigenous and exotic trees and varieties of grasses were planted on those sites to enrich vegetation cover. Indigenous trees included Olea africana, Cordia africana, and Juniperus procera; exotic tree varieties included Grevillea robusta, and acacia, while grasses that were planted included Rhodes grass (Chloris gayana), elephant grass (Cenchrus purpureus), and phalaris (Phalaris arundinacea). The result could be further refined if clearer sky satellite images were available during the main rainfall season. Figure 5. Average NDVI in Feresmay watershed during wet and dry seasons, before and after interventions (2008 threshold year).

Figure 6 illustrates the change in vegetation greenness in the treated subareas during the historic droughts of 1984, 1989, 1990, 2009, 2013, and 2015. 1984, 1989, and 1990 drought events occurred before the interventions in this watershed, whereas 2009, 2013, and 2015 drought years were after the interventions. The comparison of the average NDVI during those shock years depicts the improved resilience of vegetation greenness after treatment. Area closure improved the vegetation greenness by 26% during the drought years, while irrigation intervention improved the greenness in the buffer area by 41%, as expected, and the role of irrigation is to provide water

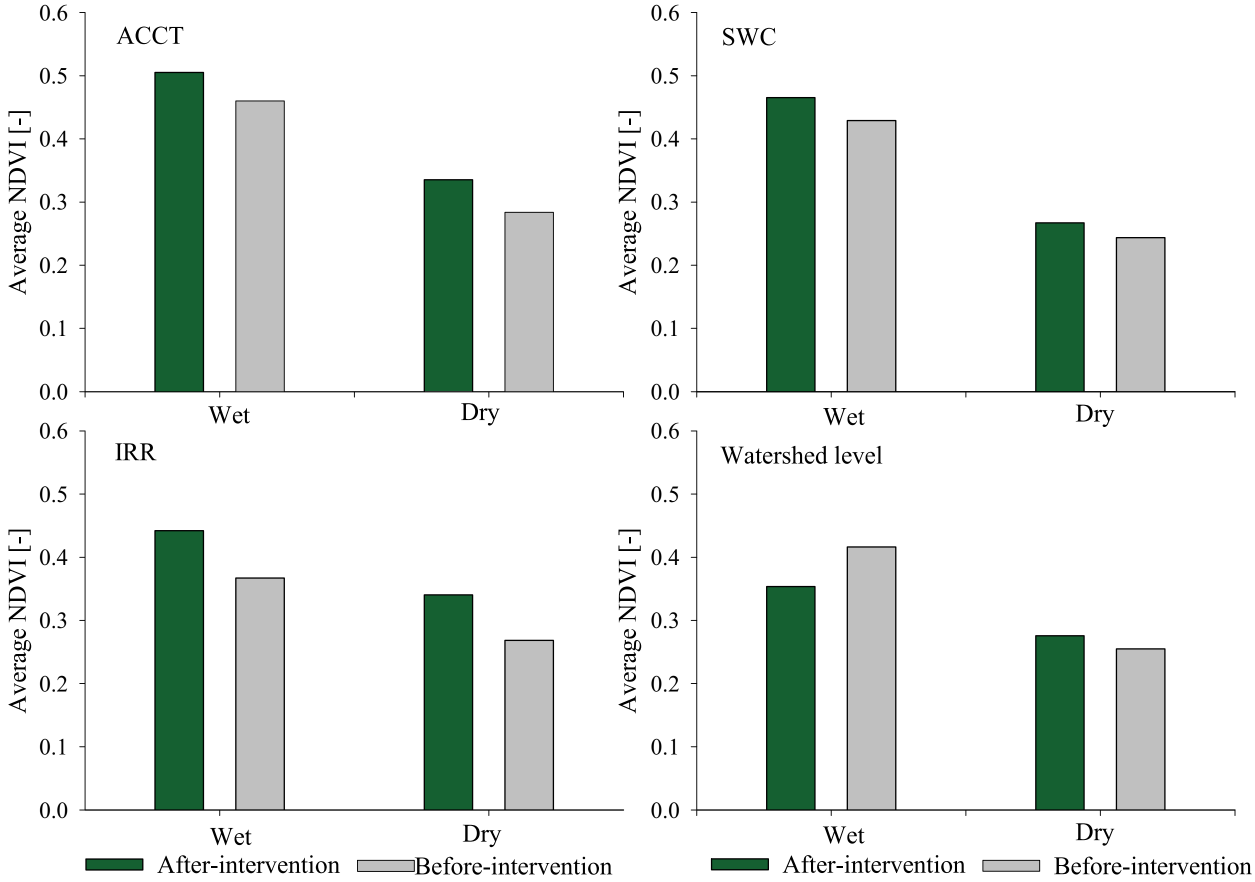

Figure 6. Average NDVI in Feresmay watershed during wet and dry seasons, before and after interventions (2008 threshold year).

during shock years to minimise crop failure due to water stress. The benefits of the interventions are reflected at the watershed scale in terms of the enhanced drought tolerance of the vegetation. Since the area closure is treated with enrichment plantations, local farmers have benefited from improved fodder and honey production in addition to the environmental and ecosystem benefits during drought years. Treatment may have improved the resilience of the regenerated vegetation in the area closure under abnormally dry conditions and augmented the vegetation's tolerance level; improvements may have been due to enhanced physiology as well as to the increased moisture-holding capacity of the soil during shock years.

Based on key informant surveys conducted with members of local communities, Gebregziabher et al. (2016) reported that area closure and plantation improved forage production in the vicinity of watersheds during dry seasons and shock years. The intervention, therefore, brought about a positive impact in the form of increased drought resilience, with vegetation in the study watersheds being able to withstand and minimise the adverse impacts of prolonged drought conditions.

3.2. Soil loss estimation and change detection using RUSLE empirical equation

Table 6 illustrates the main land use classes in the watershed. In general, positive change/improvement was observed for almost all land use types, with changes in shrub and bare lands being more pronounced. Agricultural land is the dominant land use type that covered more than 60% of the study watershed. Other studies (e.g. Alemayehu et al., 2009;

Table 6. Land use types and percentage coverage for 2019 and 2006 of the Feresmay watershed.

	Area coverage (ha)		Percent land use to the watershed area	
Land use type	2019	2006	2019	2006
Cultivated land	5195	4839	67.47	62.84
Forest and woodland	403	411	5.24	5.34
Shrub and grassland	1748	1428	22.7	18.54
Open grassland	66	65	0.86	0.85
Bare land	227	937	2.95	12.17
Built up area	36	10	0.47	0.13
Water bodies	24	10	0.31	0.13
Total	7700	7700	100	100

Nyssen et al., 2008) also reported that agriculture is the dominant land use practice in the study region. Agricultural land expanded by 4.6% in the past decade. Agricultural land use expansion is a global phenomenon driven by multiple factors such as an increase in population, food demand, and lifestyle change like increased consumption of animal products, and so on. Shrub and grassland increased by approximately 4%, whereas bare land decreased by 9%. Overall, the increases in shrub and grassland, open grassland, bare land, and waterbodies are all attributable to the expansion of watershed interventions. Such land-use changes may be partly driven by watershed conservation and irrigation interventions as verified by the implementer in the study watershed. Other studies such as Alemayehu et al. (2009) similarly reported an increase in forested land due to the implementation of area enclosure with enrichment plantations.

Figure 7 illustrates the distribution of the annual soil loss in the Feresmay watershed and the difference in soil erosion between 2019 and 2006. RUSLE was used in this section to estimate average annual soil loss values for the different slope categories. The values of the difference map (Figure 7c) were grouped into three categories to represent areas with no significant change (–0.5 to 0.5 tons $ha^{-1}yr^{-1}$), decrease (less than –0.5 tons $ha^{-1}yr^{-1}$) and an increase (greater than 0.5 tons $ha^{-1}yr^{-1}$) in the annual soil loss. The result indicated that 16% of the watershed area showed a reduction in annual soil loss, while an increase in annual soil loss has been observed in 10% of the watershed. Large parts of the watershed (74%) showed no significant change in annual soil loss.

Annual soil loss estimates in 2019 (a) and 2006 (b) representing after and before watershed conservation and irrigation interventions, while (c) shows differences in annual soil loss between 2019 and 2006 land use map.

The summary of the percentage reduction in annual soil loss for the interventions and at the watershed level is shown in Table 7. There was a 23% reduction in soil loss in areas where SWC was implemented. While 13- and 7-percent reductions were perceived in ACCT and IRR sites, respectively. These results somehow reveal the positive impacts of the interventions to minimise the annual soil loss; however, it also prompted the need for further interventions to reduce soil loss to acceptable levels. Augmenting the field-level data and process-based modelling may further improve the results in future studies.

3.3. *Soil loss estimation and change detection using SWAT*

Figure 8 shows the resulting simulated sediment yield before (Figure 8a) and after (Figure 8b) the

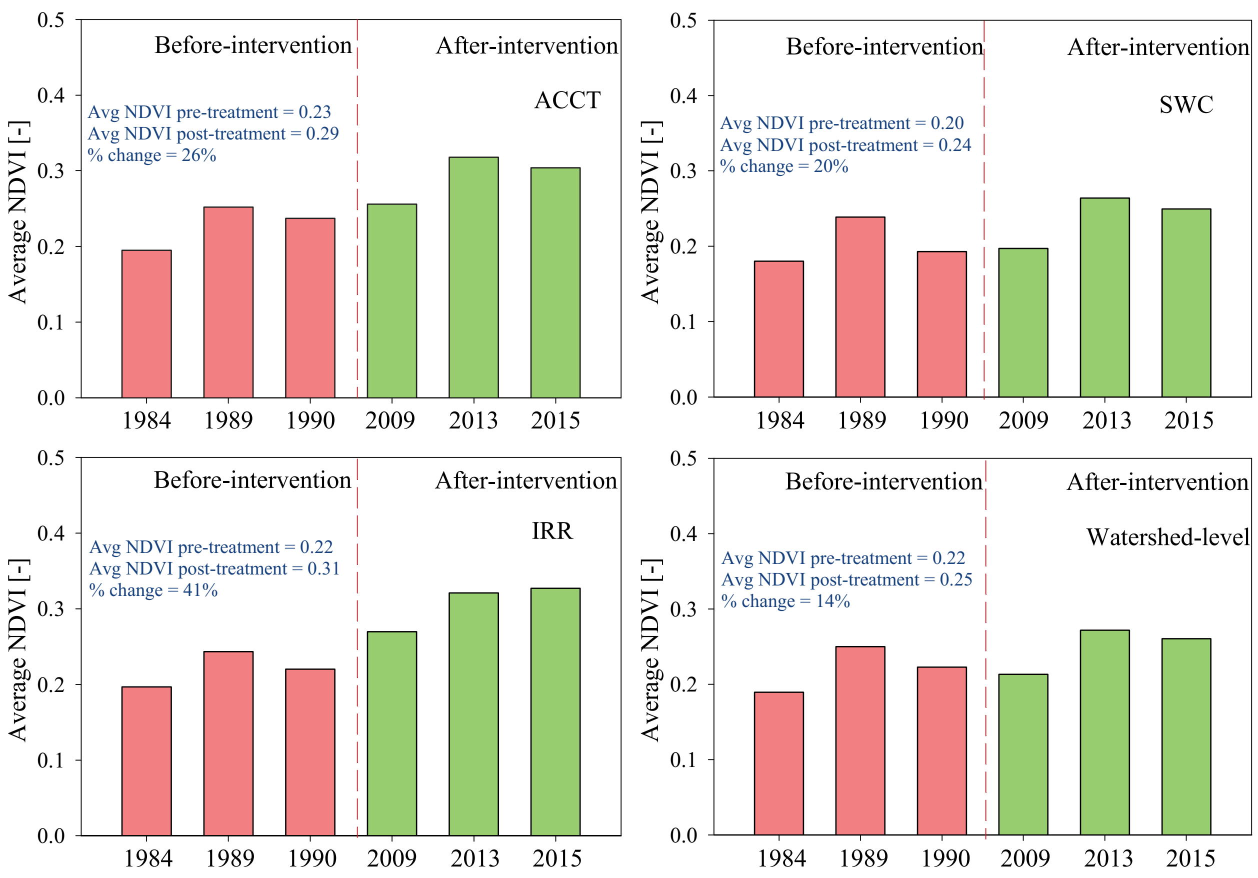

Figure 7. Average NDVI in the treated subareas of the Feresmay watershed during drought years, before and after interventions.

Table 7. Percentage reduction in soil loss due to the watershed conservation and irrigation interventions compared to baseline implementation after 2019.

SNo	Type of interventions	Area (ha)	Reduction in soil loss (%)
1	ACCT	560	13
2	IRR	308	7
3	SWC	391	23
4	Watershed-level	7700	16

intervention at the subbasin scale annual average output of the SWAT model. Similarly, the difference map (Figure 8c) represents the change in sediment yield after and before the interventions, in which negative values show a reduction in soil loss. The simulated sediment yield varies from negligible to the maximum range of 45 tons $ha^{-1}yr^{-1}$ in the study watershed (Figure 9). The average areal values of 12 and 9 tons $ha^{-1}yr^{-1}$ have been observed before and after the intervention, respectively, with an overall reduction of 3 tons $ha^{-1}yr^{-1}$ soil erosion due to the interventions. The comparison of the soil loss estimates with a previous study by Taye et al. (2015) showed that the soil loss values are within the acceptable range. Taye et al. (2015) used 21 large runoff plots to measure soil loss during the main rainfall season, which accounted for more than 80 percent of the annual soil loss in the Mayleba watershed. The watershed is located in the same agroecological zone as the study watershed, and the authors reported an average annual soil loss ranging from 11 to 14 tons $ha^{-1}yr^{-1}$. Another study by Girmay et al. (2009) also demonstrated an average soil loss ranging between 6 and 14 tons $ha^{-1}yr^{-1}$ across different land uses in Gurm Selassa and Maileba watersheds in the Tigray region. The data was collected in a total of 16 runoff plots for different land uses including cultivated land, grazing land, exclosure, and plantation area. A comparison of the model estimates of the average soil loss of the SWAT model is within the range of the measured soil loss from the previous studies; therefore,

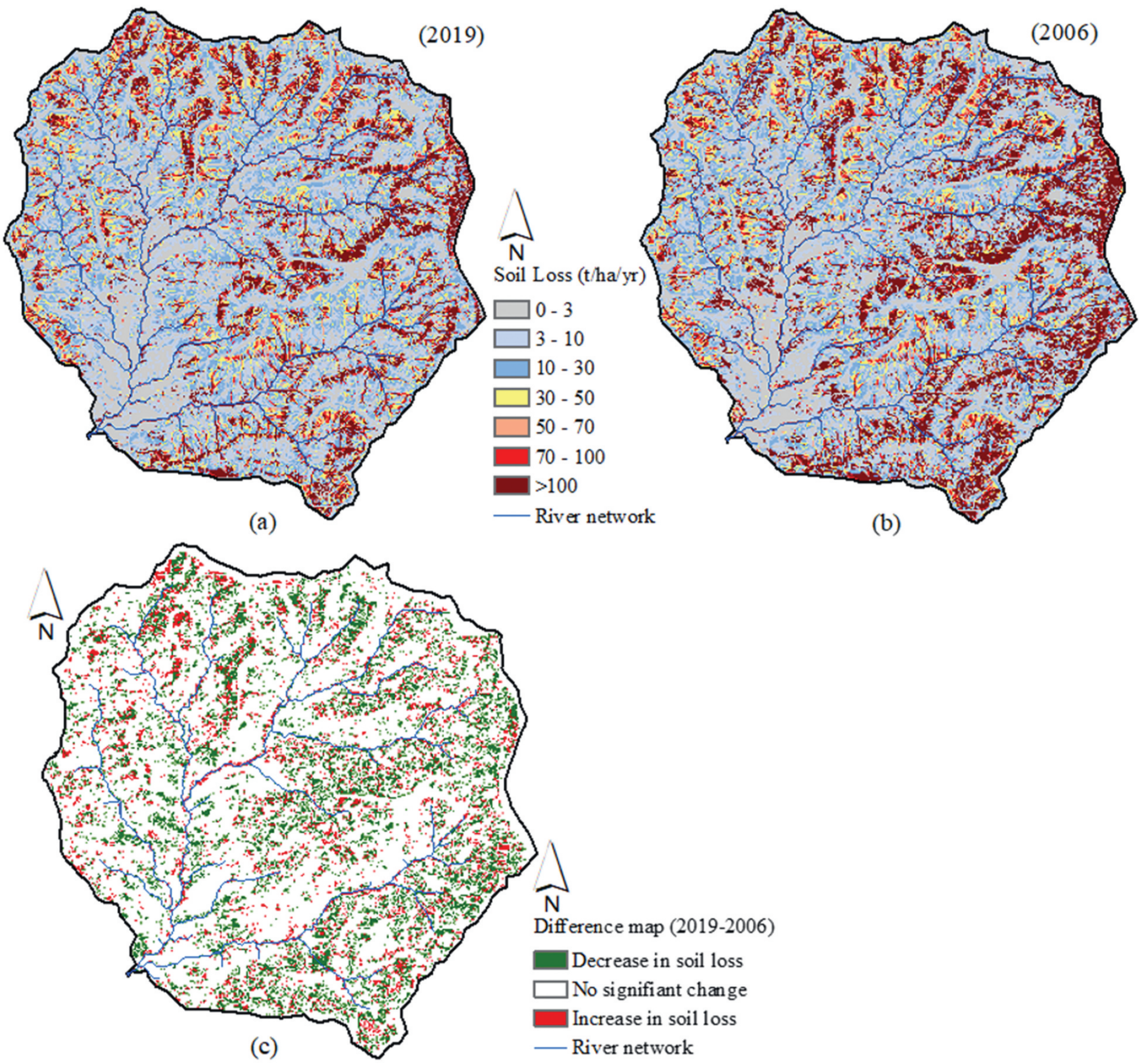

Figure 8. Annual soil loss estimates in 2019 (a) and 2006 (b) representing after and before watershed conservation and irrigation interventions while (c) shows difference in annual soil loss between 2019 and 2006 land use map.

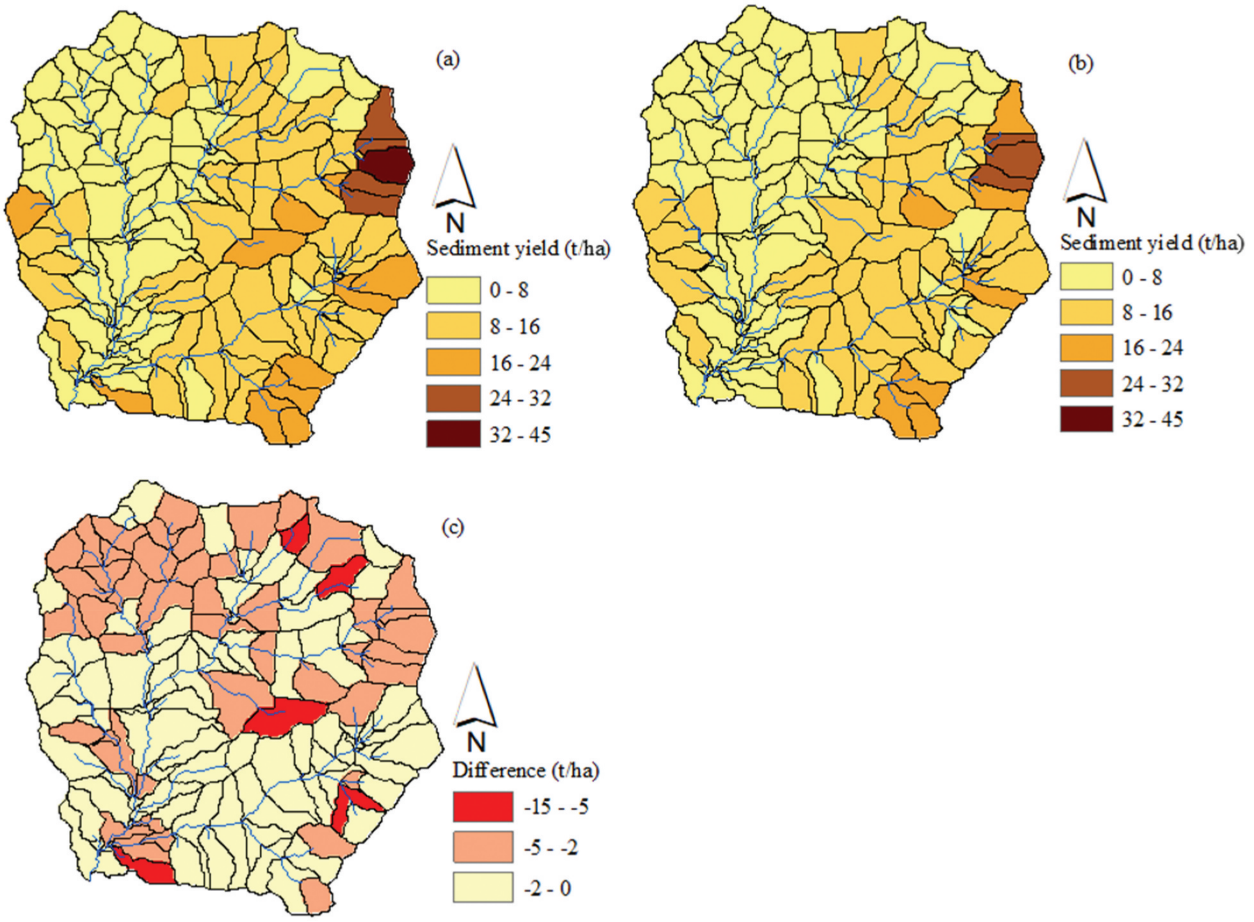

Figure 9. Simulated sediment yield before (a), after (c) the interventions, and difference map (c) in the study watershed.

the model reasonably estimated the annual soil loss in this study. The spatial patterns of the difference map revealed a decreasing trend of the sediment yield in the subbasins located mainly in the upstream parts of the watershed where some of the soil and water conservation activities were implemented. In general, 0 to 5 tons $ha^{-1}yr^{-1}$ sediment reduction has been observed in most of the upstream subbasins, while significant soil loss reduction of 5 to 35 tons $ha^{-1}yr^{-1}$ has been observed in a few subbasins. Thus, the model result of the soil loss also supports the positive impacts of the soil and water conservation interventions, although a detailed analysis is required to assess the impacts on the other water balance components.

3.4. Discussion

Area closure combined with catchment treatment was one of the most effective watershed interventions to rehabilitate degraded landscapes. In addition to reversing the trend of further degradation due to human and livestock interference, interventions such as enrichment planting of multipurpose trees and different grass species provided multiple benefits, including intercepting raindrop impact, slowing runoff speed, and providing fodder for livestock through cut and carry systems as well fodder for bees – thus enhancing honey production. Comparison of post- and pre-intervention analysis revealed up to 26% improvement in vegetation greenness in parts of the watershed receiving area closure as treatment. Other studies (e.g. Mekuria et al., 2007) have also reported the positive impacts of exclosures with respect to restoring degraded land, increasing soil fertility, and declining soil erosion. Furthermore, vegetation enhancement was found to be even more drastic in areas where area closure was combined with other SWC interventions such as terraces, soil and stone bunds, and gully rehabilitation structures. Those structures are not only believed to have benefited from intercepting the run-off flood but also retained it for longer durations to enhance infiltration into the soil profile, which in turn is likely to provide excess water for plant growth. This study also demonstrated that small-scale irrigation interventions enhanced vegetation greenness largely during the dry seasons in the buffer zone. The irrigation interventions improved vegetation greenness in the buffer area by 26% and 19% in the dry and wet seasons, respectively. This finding was expected, as irrigation water is applied to grow crops in the dry season either through bridging rainfall variability for the rainfed crops or meeting the crop water requirements for a full crop cycle. Moreover, the irrigation water application has a spillover effect on the surrounding fields and contributes to the improvement of vegetation greenness for the watershed level analysis. Moreover, the application of irrigation water enhances the residual soil moisture in the irrigated fields that support vegetation growth during the fallow period. The farmers in the highlands of Ethiopia, such as in Tigray, often treat their land with SWC structures, such as soil bunds, terraces, and contours. Many farmers also plant grasses on the risers of these physical structures to maintain their stability of the structures. Although such vegetation cover is intended to provide stability to the structures, it also enhances vegetation greenness and provides fodder for livestock. In general, watershed conservation and irrigation interventions were found to benefit the local community by improving agricultural productivity, reducing soil erosion, enhancing soil moisture and vegetation greenness, and thus improving the livelihoods of smallholder farmers.

Land use change is occurring in most parts of Ethiopia, especially in the Northern Highlands, where there is a high population increase and agricultural expansion to produce more food (Mustard et al., 2012). This study also found that, in the Feresmay watershed, agricultural lands have expanded by about 5% (Table 8) between 2006 and 2019. An increase in the waterbodies was also observed, which may be due to the construction of water storage structures (e.g. mini dams) to supply water for irrigation and emerging wetlands in the flat plain due to the interventions implemented in the upstream part of the watershed as confirmed by the implementers. Authors such as Belay et al. (2015) and Alemayehu et al. (2009) have also reported the expansion of cultivation land and the built-up area in large parts of the study watershed in the Northern Highlands. The land use conversion in the past few decades and other anthropogenic activities such as deforestation, conversion of shrubs and woodlands to agricultural land, and intensive cultivation aggravated soil erosion and land degradation (Gebresamuel et al., 2010). Several other studies also indicated that cultivated land is the dominant land use in many watersheds in the Ethiopian highlands and covers more than 50% of the total area of the watersheds (Alemayehu et al., 2009; Gebresamuel et al., 2010; Nyssen et al., 2008), which is in-line with the land use classification result obtained in this study (Table 8). Although agricultural land is the dominant land use type in the study watershed, Alemayehu et al. (2009) indicated a reduction in intensively rainfed cultivated land by 3.4% and an increase in irrigation cultivation lands (1.1%) between 1994 and 2005. Gebresamuel et al. (2010) reported a reduction in cultivation land by 5.5 ha yr^{-1} in one of the watersheds considered in their study, even though forests and woodlands completely vanished from the watersheds. The reduction in agricultural land may be due to the expansion of other practices such as settlement due to population growth.

As expected, the rate of soil erosion in the watershed increased with increase in the slope gradient. Table 8 summarises annual average soil losses corresponding to the different slope categories classified based on FAO slope classes. The lowest average annual soil of 4 tons $ha^{-1}yr^{-1}$ was simulated on the flat to gentle (0–3%) slope range, while the highest average annual soil loss value of 39 tons $ha^{-1}yr^{-1}$ was obtained from the extreme steep slope range of above 35%.Average soil loss values simulated on slope classes 2, 3 and 4 were 11 tons $ha^{-1}yr^{-1}$, 20 tons $ha^{-1}yr^{-1}$, and 28 tons $ha^{-1}yr^{-1}$, respectively. Most of the structural watershed conservation interventions were implemented in the highlands where the slope is steep, and soil is prone to severe erosion. Run-off plot level studies (Nyssen, Clymans, et al., 2009; Nyssen, Poesen, et al., 2009; Taye et al., 2013) as well as RUSLE-based modelling estimates (Brhane & Mekonen, 2009; Haregeweyn et al., 2012) in the same study region reported average annual soil losses ranging from 3.5 to 39 tons $ha^{-1}yr^{-1}$, which are in-line with the values estimated in this study (Table 8).

A comparison of simulated soil losses for the corresponding slope categories under 'treatment' and 'without treatment' conditions revealed 5% to 22% reduction in average annual soil losses under 'treatment' conditions. Notably, Gebrernichael et al. (2005) indicated a 68% reduction in average annual soil loss using qualitative and quantitative approaches in 202 run-off plots treated with soil bunds. The substantial difference in the effectiveness of soil bunds in the 2005 study versus this study could be explained by the fact that soil bunds are often built on croplands where erosion rates on untreated plots are generally expected to be high due to repeated ploughing and loosening of the soil. In addition, the rate of soil erosion is expected to be generally higher in less degraded lands as compared to severely degraded ones like the study watershed. Because there is less to be eroded in severely degraded lands, which is most likely the case of the watershed considered in this study. The fact that soil bunds or any other barrier, for that matter, are significantly impacting moderate slopes

Table 8. Annual soil loss at the corresponding FAO slope classes.

Class	Slope gradient (%)	Annual soil loss (tons $ha^{-1}yr^{-1}$)	Description of slope
1	0–3	4	Flat to gentle
2	3–12	11	moderate
3	12–20	20	steep
4	20–35	28	Very steep

with deep cultivated soils could also be the case in most of the 202 run-off plots that Gebremichael et al. (2005) considered in their study. Since the RUSLE model applied in this study estimates sheet and rill erosion and ignores the effects of gully erosion and dispersive soils (Renard et al., 1991), the result presented in this study might underestimate soil erosion.

4. Limitations and future recommendations

In this study, field-measured data at the plot level or the outlet of the watershed was not carried out due to the security situation in the Tigray region, even though comparing the model estimate with ground truth observation was one of the activities initially planned to be completed in this study. Future studies using field measurement data may improve the accuracy of the finding of this study. Estimates using similar methods as in this study in the future would benefit from access to more precise information on the implementation years and the use of more cloud-free imageries during the rainy seasons. Moreover, ground confirmation of the remotely sensed information is expected to highly improve the precision of the model estimates.

5. Conclusion

The trend analysis illustrated that there was no significant change in rainfall after the interventions as compared to before the interventions period. Therefore, change in rainfall is not a substantial reason for the change in vegetation greenness or other biophysical variables in the study watershed. The findings of this study further indicated positive impacts of the watershed-conservation and irrigation interventions in terms of enhancing vegetation greenness and reducing soil loss, although the effectiveness varies by intervention. The interventions also contributed to enhanced availability and prolonged duration of soil moisture, as evidenced by the higher NDVI values for the dry periods compared to the without-treatment conditions. The study also highlighted that those investments in watershed conservation and irrigation interventions might have contributed to the improvement of the livelihoods of smallholder farmers in the watershed. However, that would need further confirmation through ground-level socio-economic assessments. While the results of this study suggested that investments in watershed conservation and irrigation interventions generally have a positive impact on communal assets including natural resources like soil, water, and vegetation, as well as socio-economic and environmental factors that enhance resilience and overall community wellbeing, further research using an integrated approach including remote sensing, erosion modelling, on-site assessment, and household, and community level socio-economic and environmental surveys may offer a more complete understanding of the holistic impacts of these interventions. Additionally, experience and previous assessments have shown that the effectiveness of such interventions decreases with service life, suggesting that regular maintenance is necessary to maintain their effectiveness.

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Disclaimer

The views expressed in this manuscript are solely those of the authors and not representing any of the respective agencies.

Data availability statement

Data sharing is not applicable to this article, as we mainly used public-domain secondary datasets that are freely accessible.

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Appendix

Appendix A: Trends analysis of rainfall before and after the intervention

Table A1 summarises monthly, seasonal and annual rainfall before and after the interventions. The result shows there was no significant change in the average rainfall and percent of rainfall before and after the interventions during kiremt and at annual time scales. Conversely, relatively higher values of the standard deviation were observed before the intervention compared to after the interventions during the kiremt and annual time scales. This might be justified with the occurrence of extreme historic drought events before the intervention period (e.g., 1984 and 1991). The coefficient of variation (CV), which captures the rainfall variability, illustrates less variability of the seasonal and annual rainfall before and after the interventions except for the kiremt season that shows moderate variability before the interventions as per the rainfall variability classification suggested by Hare (1983). Hare (1983) classified the rainfall variability based on CV values as low (<20), moderate (20-30) and high (>30). CV values greater than 100% were observed during the dry months (December to February) both before and after interventions illustrating the sensitivity of CV for rainfall pattern during the dry months.

Figure A1 depicts the trends in seasonal rainfall before (left) and after (right) the interventions for each aggregated group of interventions. Overall, there is no significant change in the observed rainfall after the intervention. While an upward increasing trend in the treated subareas was observed as compared to before the intervention. The comparison of the slope of the trendlines shows that there is a relatively smaller slope value after the intervention period as compared to before.

Table A1. Summary of the long-term average monthly, seasonal, and annual rainfall before (B) and after (A) the interventions in Feresmay watershed.

	Average rainfall (mm)		Percent of rainfall (%)		Standard Deviation		CV (%)		Skewness	
Months	B	A	B	A	B	A	B	A	B	A
January	0.7	0.4	0.1	0.1	1.2	0.9	>100	>100	1.9	2.3
February	1.9	1.7	0.3	0.3	2.5	3.3	>100	>100	1.3	1.8
March	32.0	19.7	5.1	3.1	16. 8	6.6	52	34	0.9	−0.3
April	55.6	57.7	8.8	9.1	29.1	24.5	52	43	0.8	2.0
May	49.6	54.7	7.9	8.6	32.6	28.6	66	52	1.7	0.4
June	53.7	49.9	8.5	7.9	25.3	17.5	47	35	1.9	1.0
July	187.9	194.7	29.8	30.7	55.0	40.0	29	21	0.6	0.1
August	191.7	183.7	30.4	29.0	53.2	37.5	28	20	−0.3	0.0
September	30.0	35.7	4.7	5.6	12.2	14.7	41	41	0.5	0.8
October	15.5	17.8	2.4	2.8	10. 5	7.3	68	41	1.4	0.2
November	11.5	17.7	1.8	2.8	6.6	11.6	58	65	2.0	1.3
December	1.4	0.5	0.2	0.1	2.8	1.5	>100	>100	2.0	3.4
Kiremt	463.21	464.0	73.4	73.2	100.9	56.2	22	12	0.4	−0.4
Annual	631.26	634.1	100.0	100.0	95.4	64.9	15	10	−0.6	−0.7

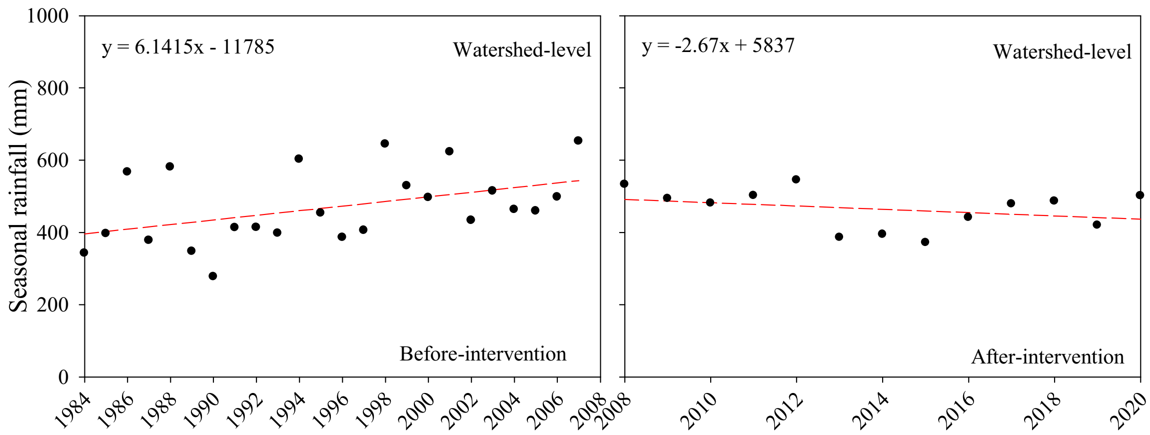

Figure A1. Trend analysis of the seasonal/kiremt rainfall before (left) and after (right) the interventions for watershed level analysis.

Table A2. P-values of the modified Mann – Kendall non-parametric trend analysis of the rainfall before and after the interventions in Feresmay watershed.

Periods	Monthly	Seasonal	Annual
Before intervention (1982–2007)	0.77	0.02	0.23
After intervention (2008–2020)	0.51	0.56	0.34

Non-parametric Mann–Kendall trend analysis of the monthly, seasonal, and annual rainfall was also analysed before and after the interventions. No significant trend was observed in monthly and annual rainfall at 95% ($\alpha = 0.05$) level of significance (Table A2) before and after the interventions. P-values greater than 0.05 have been indicated during monthly and annual time periods. Similarly, no trend was identified for seasonal rainfall after the interventions, though a significant trend (p-values<0.05) was observed before the interventions. Overall, the result of the trend analysis illustrates no notable changes in rainfall after the interventions as compared to before the interventions. Therefore, changes in precipitation are not a substantial reason for changes in vegetation greenness and other biophysical variables.