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Original Research

Effects of land management practices on runoff and soil and nutrient losses in the rainfed agroecosystem of the Beles River Basin, Ethiopia

Yenesew Assaye [a, b, *], Gizaw Desta [c], Eyayu Molla [a], Zenebe Adimassu [d]

[a] Department of Natural Resources Management, College of Agriculture and Environmental Sciences, Bahir Dar University, Bahir Dar P.O. Box 79, Ethiopia
[b] Natural Resources Management Research Directorate, Ethiopian Institute of Agricultural Research (EIAR), Addis Ababa P.O. Box 2003, Ethiopia
[c] International Crops Research Institute for the Semi-Arid Tropics (ICRISAT), Addis Ababa P.O. Box 5689, Ethiopia
[d] International Center for Tropical Agriculture (CIAT), Addis Ababa P.O. Box 5689, Ethiopia

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A B S T R A C T

The Beles River Basin is facing severe soil erosion driven by human-induced activities, leading to significant losses of soil organic carbon (SOC) and nutrients (nitrogen (N) and phosphorus (P)). Effective land management practices (LMPs), including mechanical, biological, and agronomic techniques, are potential strategies for mitigating this degradation, but their effectiveness depends on site-specific and agroecological conditions. However, limited information is available on this aspect of the study area. The objective of the current study was to evaluate the effects of LMPs in the warm subhumid lowlands of the Beles River Basin on runoff, soil loss, and sediment-associated losses of SOC, N, and P from agricultural land. Four LMPs (vetiver grass strips (VGS), conservation agriculture (CA), soil bunds (SB), and fanya juu (FJ)) were evaluated via runoff plots arranged in a randomized complete block design (RCBD) with three replicates. Farmer practices were used as a control (C). The experiments, which were performed over three years (2021–2023), generated runoff, soil loss, and nutrient loss data. The three-year mean annual runoff ranged from 58.5 to 407.5 mm, and the soil loss ranged from 4.3 to 45.4 t/ha, whereas the annual rainfall varied between 1,402 mm in 2021, 1,254 mm in 2022, and 1,261 mm in 2023. On average, runoff was reduced by 36%–85%, and soil loss was reduced by 53%–91% in the LMP-treated plots. Additionally, sediment-associated losses of SOC, N, and P were reduced by 55%–90%, 52%–90%, and 28%–72%, respectively. The results revealed significant differences ($p < 0.05$) among the treatments in terms of reducing runoff, soil loss, and sediment-associated losses of SOC, N, and P. The mean annual runoff and soil loss rates during the study were 407.5, 230.3, 136.3, 59.6, and 58.5 mm and 45.4, 21.5, 11.1, 4.5, and 4.3 t/ha under the control, VGS, CA, SB, and FJ practices, respectively. The highest rates of runoff and soil loss were observed under the control conditions (407.4 mm and 45.4 t/ha). Runoff, soil loss, SOC, and nutrient (N and P) losses were significantly lower ($p < 0.05$) in the plots treated with FJ and SB than in the other plots. However, CA and VGS also significantly varied ($p < 0.05$) in reducing runoff, soil, SOC, and nutrient losses over the years. These results highlight the key role of LMPs in warm subhumid lowland rainfed agroecosystems as effective land management techniques for controlling soil and nutrient loss.

1. Introduction

Accelerated soil erosion by water is a major global challenge contributing to land degradation (Bouamrane et al., 2024; FAO, 2019; Fekadu et al., 2024; Fenta et al., 2021; Lal, 2014; Meshesha et al., 2013; Montanarella et al., 2016). The Global Assessment of Land Degradation (GLADA) by the United Nations Environment Programme (UNEP) revealed that soil erosion by water affects approximately 1.1 billion ha of land (Bai et al., 2008). Among the affected continents, Asia (48%) was the most damaged, followed by Africa (21%), Latin America and the Caribbean (15%), Europe (10.5%), Australia and the Pacific (5.5%), and North America (5%). Consequently, the annual cost of land degradation was estimated at nearly $231 billion (Nkonya et al., 2016). In sub-Saharan Africa (SSA), 65% of the land has been affected by degradation, resulting

* Corresponding author. Department of Natural Resources Management, College of Agriculture and Environmental Sciences, Bahir Dar University, Bahir Dar, Ethiopia.
E-mail address: yenesewassaye@gmail.com (Y. Assaye).
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in annual grain yield losses of approximately 280 million t (Wolka et al., 2018), with associated costs amounting to $9.3 billion (Chinzila, 2017). Ethiopia, located in SSA, is also impacted by land degradation, with annual costs of $4.3 billion (Gebreselassie et al., 2016).

In Ethiopia, accelerated soil erosion has affected both onsite agricultural productivity decline and the offsite rapid siltation of reservoirs (Ayele et al., 2021; Haregeweyn et al., 2006; Lemma et al., 2018). This accelerated erosion, driven by human activities such as the conversion of natural ecosystems into agroecosystems, deforestation, overstocking, overgrazing, and unsustainable farming practices, is a growing concern (Yasir et al., 2014; Fitawok et al., 2020; Oljirra, 2019; Sonneveld et al., 2010; Tadesse & Ahmed, 2023). As a result, the nation's annual gross soil loss due to water erosion is estimated at 1.9 billion t per year (Fenta et al., 2021), with mean soil loss rates ranging from 50 to 140 t/ha/y and sometimes reaching 300 t/ha/y (Adimassu et al., 2017; Desta et al., 2021a). Thus, soil losses exceeded the mean soil loss tolerance rate of 10 t/ha, as suggested by Hurni (1983) for Ethiopian conditions. Accelerated soil erosion also depletes soil organic carbon and essential plant nutrients, resulting in national-level nutrient depletion rates estimated at 3,700, 122, 13, and 82 kg/ha/y for SOC, N, P, and K, respectively (Haileslassie et al., 2005; van Beek et al., 2018). Consequently, Ethiopian agricultural productivity is generally substantially below global and regional standards, as reported by Tamene et al. (2017).

Among the various land uses in Ethiopia, soil erosion by water is most severe on agricultural land (Fenta et al., 2021; Haregeweyn et al., 2017; Meshesha et al., 2013). The mean soil loss on agricultural land in Ethiopia is estimated to range between 50 and 140 t/ha (Desta et al., 2021a). Furthermore, Fenta et al. (2021) reported that agricultural land experiences an annual soil loss of 949 million t, which constitutes more than 50% of the country's total soil loss and has lasting effects on land use.

The Abay River Basin experiences the highest degree of soil erosion in Ethiopia, with an average loss of 46 t/ha/y and a total annual loss of 573 million t (Fenta et al., 2021; Haregeweyn et al., 2017; Yasir et al., 2014). This incoming sediment also poses a significant risk to the Grand Ethiopian Renaissance Dam (GERD), potentially reducing its ability to generate electricity in the short to medium term. In addition, the soil loss Basin-level nutrient export rates are estimated to be 1.89, 0.17, and 0.62 million t/y for SOC, N, and P, respectively (Haregeweyn et al., 2017). This combined loss of soil and nutrients leads to a decline in agricultural productivity, which in turn affects socioeconomic aspects.

To combat land degradation, the government of Ethiopia, in collaboration with donor agencies, initiated soil and water conservation (SWC) programs in the 1970s and 1980s, with a focus on food-for-work projects in drought-prone areas (Haregeweyn et al., 2015; Herweg & Ludi, 1999). This program continued until it was replaced by a participatory community-based watershed development approach in the early 1990s (Haregeweyn et al., 2012). To further strengthen the existing program and address previously unaddressed areas, the nation-wide Sustainable Land Management Program (SLMP) was launched in 2008 (Schmidt & Tadesse, 2017). As a result, in the Ethiopian highlands, 7.7 million ha (23% of the area requiring restoration) have been covered by land management measures (Bantider et al., 2019). Despite long-standing efforts and investments in land restoration by the government and multi-stakeholder collaborations, many findings conclude that the challenge of soil erosion continues due to a lack of implementation of appropriate and site-specific practices (Abera et al., 2020; Biratu et al., 2023; Desta et al., 2021a).

Several studies have indicated that soil erosion and conservation research have been extensively conducted in the highlands of Ethiopia, particularly in the northern, northwestern, central, southern, and rift valley areas (Abera et al., 2020; Biratu et al., 2023; Desta et al., 2021a; Fenta et al., 2024). Conversely, in the lowlands of Ethiopia, research on soil erosion and conservation has received less attention than in the highlands. This is due to perceptions related to lower population density and low pressure on land, lower annual rainfall, and less favorable conditions for crop production. However, in low-altitude areas with high rainfall and temperatures, particularly the Beles River Basin (Ali et al., 2024; Belay et al., 2019), studies addressing soil erosion, nutrient loss, and soil erosion control techniques in these areas are limited (Nyssen et al., 2018).

The Beles River Basin is recognized by the Ethiopian government as a basin with tremendous potential for contributing to national economic growth (Assaye et al., 2024; World Bank, 2008). Consequently, several megaprojects have been running in the basin, including the GERD (Teklemariam et al., 2017), the Integrated Beles Sugar Development (Fantini et al., 2018), the Tana-Beles Hydro Power Project (Annys et al., 2019), the Tana Beles Resettlement Program (Abbute, 1997), and mechanized farm investments (Teklemariam et al., 2017). However, these development contributors have led to significant challenges in the basin, including land degradation in the form of soil erosion by water and the deterioration of ecosystem services (Abebe et al., 2021; Annys et al., 2019; Nyssen et al., 2018). In addition, following the 1984 and 1985 drought and famine, many people from the northern and southern parts of the country resettled in this basin (Woldemeskel, 1989). To secure livelihoods for settlers, extensive mechanized farming, deforestation, infrastructure development, and the use of synthetic agricultural chemicals, which deplete natural resources without implementing environmental sustainability measures, are needed. Nyssen et al. (2018) highlighted that land use and management are critical research areas that remain inadequately addressed. Teklemariam et al. (2017) conducted a field survey of 20 catchments within the basin and reported a mean suspended sediment concentration of 6.44 g/L. The World Bank (2008) indicated that rainfall-runoff models for this basin show a total runoff of approximately 5,690 million m^3/y, resulting in an estimated soil loss of approximately 63.6 million t of rapid siltation into the GERD.

The current study is crucial for the onsite restoration of rainfed agroecosystems and the offsite protection of the GERD from sediment loads. The GERD is situated at the outlet of the Beles River Basin, where part of the basin is occupied by GERD reservoirs, which directly contribute to siltation in the dam. Therefore, the objective of the current study was to quantify the effects of different land management practices on runoff, soil and sediment-associated SOC and nutrient losses in agricultural lands under rainfed agroecosystems. The specific objectives were (1) to assess the effectiveness of mechanical soil conservation structures (soil bunds and fanya juu), biological (vetiver grass strips), and agronomic (conservation agriculture) practices in reducing runoff and soil loss and (2) to quantify reductions in sediment-associated losses of SOC and soil nutrients (N and P) under different land management practices.

2. Materials and methods

2.1. Description of the study area

The study area is located in the rain-fed agroecosystem of the middle Beles River Basin, as shown in Fig. 1. This basin was chosen because more than 87% of the basin's agroecological setting consists of humid and subhumid lowlands (Assaye et al., 2024). The area has experienced severe land degradation due to long-term occupation by the resettlement villagization program. The farming system in this area is characterized by a mixed crop–livestock farming system, semipastoralism, and mechanized farming. The Tana-Beles Resettlement village covers an area of 220,000 ha within the basin (11.31°–11.33°N and 36.41°–36.33°E), as shown in Fig. 1. The elevation of the study area ranges from 710 to 1,300 m above sea level. The Beles River flows from north to south, crosses into the southwest, and divides the study area into two zones: left and right, as illustrated in Fig. 1. The right zone is characterized by a crop–livestock mixed and semipastoralist system, whereas the left zone features mechanized farming systems.

The long-term mean annual minimum and maximum temperatures were recorded as 16.6 and 32.7 °C, respectively (Fig. 2). The coldest months were December and January, whereas March, April, and May were the hottest (Fig. 2). The annual mean rainfall was approximately 1,600 mm, characterized by a unimodal pattern that typically extends from April to the end of October, with the highest

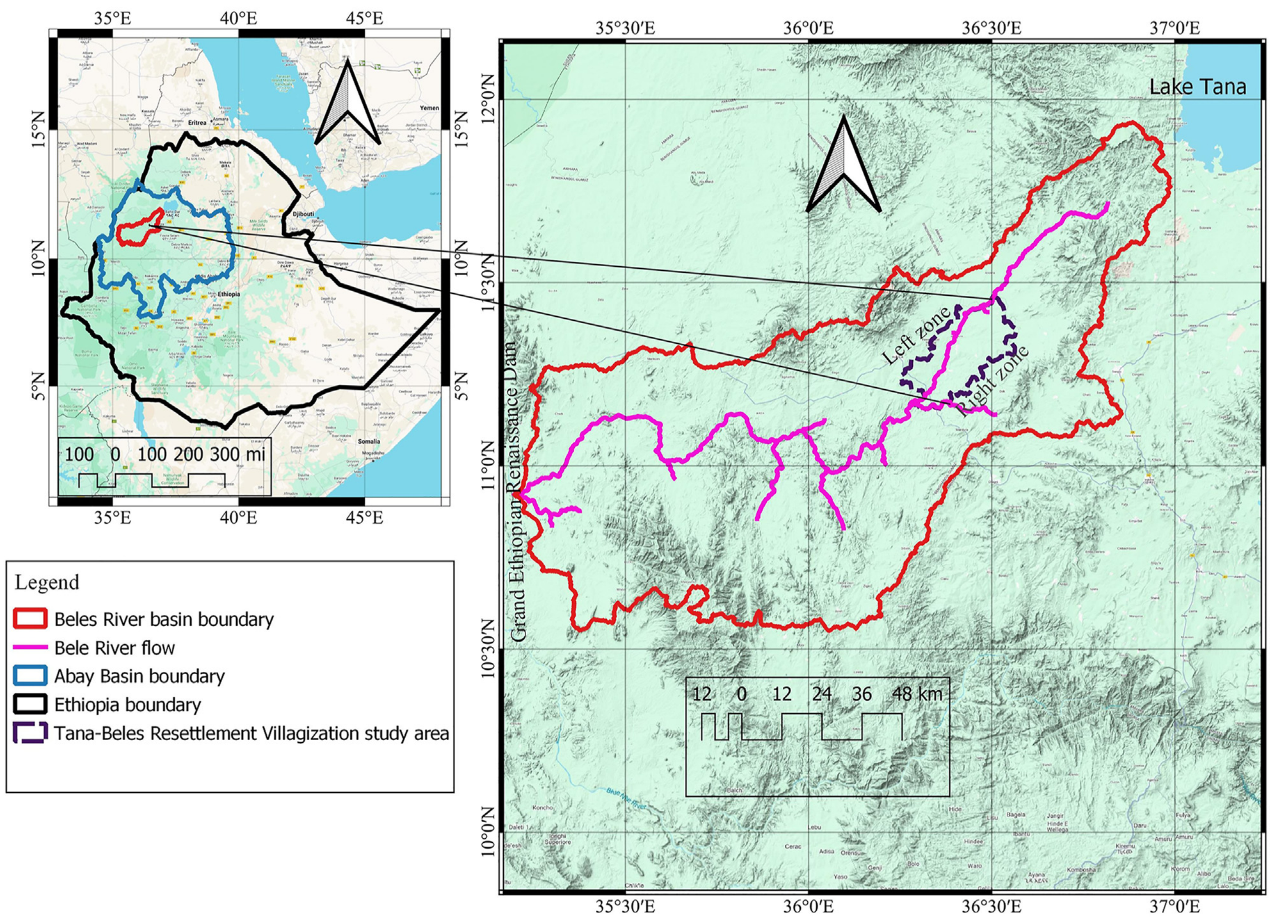

Fig. 1. Location map of the middle Beles River Basin, Ethiopia.

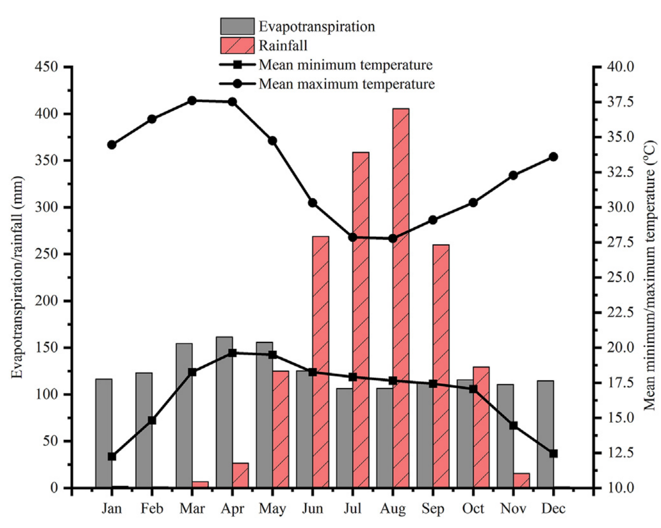

Fig. 2. Long-term climate data for the middle Beles River Basin. Data collected from Pawe Agricultural Research Center, first-class meteorological station (1997–2023).

rainfall occurring in July and August (Fig. 2). Rainfall exceeds evapotranspiration during June, July, August, September, and October, whereas evapotranspiration is greater in the remaining months (Fig. 2). The geology of the middle Beles area consists of Tertiary trap basalts and strongly folded and faulted Precambrian rocks (Nyssen et al., 2018). Additionally, thin intercalations of Mesozoic sedimentary rocks separate these two main formations (Nyssen et al., 2018).

The predominant vegetation includes dry combretum woodland with species such as Acacia, mango (*Mangifera indica*), water pear (*Syzygium guineense*), African wild olive (*Olea africana*), *Wanza* (*Cordia africana*), and neem (*Azadirachta indica*) (Nyssen et al., 2018). Traditional agroforestry practices feature lowland bamboo (Oxytenanthera abyssinica) in boundary plantations, mango with maize, and papaya with maize. The main crops grown in the study area include maize (*Zea mays*), finger millet (*Eleusine coracana*), sorghum (*Sorghum bicolor* L. *Moench*), soybean (*Glycine max*), rice (*Oryza sativa*), and groundnut (*Arachis hypogaea*). Groundnuts and soybeans are major cash crops, whereas the other crops are primarily consumed. Major livestock species include cattle, goats, poultry, and donkeys. Moreover, off-farm activities such as daily labor, petty trade, fishing, and charcoal production provide alternative sources of income.

2.2. Rainfall and soil characteristics

The total annual rainfall recorded at the experimental site was 1,402 mm in 2021, 1,254 mm in 2022, and 1,261.5 mm in 2023 (Fig. A1 in the Electronic Supplementary Material (ESM)). The proportion of annual rainfall that occurred between June and October was 88% in 2021, 95% in 2022, and 85% in 2023 (Fig. A1 in the ESM). These months were chosen because rainfall exceeded evapotranspiration (Fig. 2), potentially generating runoff in the study area. During this study period, there were 128 rainy days in 2021, 139 rainy days in 2022, and 147 rainy days in 2023. Although the total rainfall was greater in 2021, the number of rainy days was greater in both 2022 and 2023. This indicates a significant variation in rainfall and the number of rainy days over the years (Fig. A1 in the ESM). Each year, the highest rainfall occurred in July, followed by August and June (Fig. A1 in the ESM).

The highest daily rainfall amounts were recorded on June 29, 2021 (55.2 mm), July 31, 2022 (74 mm), and June 8, 2023 (69.1 mm) (Fig. A2 in the ESM). These factors clearly indicate that the extreme rainfall in June and July was linked to the area's rainfed agriculture and intensive tillage practices for seedbed preparation during these months. This combination led to significant soil erosion, which persisted until cover crops were established. Ebabu et al. (2019) reported that in the Upper Blue Nile River Basin, Ethiopia, 85% of the annual rainfall occurs during the four rainy months from June to September. Similarly, Wolka et al. (2021) reported that 83% of the annual rainfall in the Omo-Gibe River Basin in southern Ethiopia was recorded between March and October. Amare et al. (2014) reported similar patterns of rainfall onset and termination in the Debre Mewi watershed. Therefore, in the current study area and elsewhere in Ethiopia, soil erosion caused by rainfall occurs during the main rainy season (May–October).

The dominant soil groups in the study area are Leptosols, Nitisols, Luvisols, and Vertisols (Ali et al., 2024). The selected soil physicochemical properties of the experimental site are listed in Table 1. The results revealed that the soil was clay in texture and had an optimum bulk density (1.21–1.34 g/cm^3) and soil porosity (49%–54%). The combination of these properties influences soil erosion, either by promoting it within the soil profile or by being able to store large quantities through adsorption. Moreover, the soil has low P, SOC, and pH with high total N and a moderate cation

Table 1
Soil characteristics at the experimental site before the establishment of the runoff plots.

Soil property	Unit	Soil depth (cm)	
		0–20	20–40
Clay	%	44	44
Silt	%	16	14
Sandy	%	40	42
Class	Clay	Clay	Clay
Soil bulk density	g/cm^3	1.21	1.23
Soil porosity	%	54	49
Soil pH		5.43	5.44
SOC	mg/kg	1.08×10^4	1.11×10^4
Organic matter	mg/kg	1.86×10^4	1.92×10^4
Total N	mg/kg	1.4×10^3	1.64×10^3
Available phosphorus (P)	mg/kg	1.08	1.11
Cation exchange capacity (CEC)	cmol$_+$/kg	18.94	17.93

exchange capacity (CEC), as rated by Hazelton and Murphy (2007), indicating that the soil could bind water and resist erosive forces. These characteristics suggest that soil has the ability to retain water and resist erosive forces. Understanding how soil characteristics interact is crucial for predicting how much water will be absorbed by the soil and how much water will contribute to surface runoff (Qi et al., 2020). Furthermore, the dynamic relationships between soil properties and water movement and storage are key for effective water management (Fabrizzi et al., 2005). Thus, rainfall, soil erosion, and soil characteristics are intricately linked.

2.3. Experimental setup

The experiments were performed over three consecutive years (2021, 2022, and 2023) during the main rainy season. The treatments were implemented under natural rainfall conditions on agricultural land. The land management practices under evaluation were (1) conventional practices (C), (2) vetiver grass strips (VGS), (3) conservation agriculture (CA), (4) soil bunds (SB), and (5) fanya juu (FJ), and a detailed description is given in Table 2. The runoff plot was arranged in a randomized complete block design (RCBD) with three replicates, as shown in Figs. 3 and 4. This design was used to minimize variation and ensure accurate estimation of experimental errors within the units (Hudson, 1993).

Parameter data were measured via the methodology outlined by Rüttimann et al. (1995) and Wolka et al. (2021). Each hydrologically isolated runoff plot was 22.6 m in length (along the slope) and 4 m in width (along the contour line) (Figs. 3 and 4). The spacing between consecutive mechanical structures or grass strips was determined on the basis of the vertical interval, as reported by Hurni et al. (2016) for Ethiopian conditions. The average slope gradient of the runoff plots was 9%, with a uniform soil type (Nitisols) and aspect (Northwest). The experimental site is also free from the presence of rock fragment cover (Fig. 3) because it affects soil erosion. All plots were bounded by corrugated iron sheets, with 15 cm inserted into the ground and 35 cm left above the surface to prevent lateral movement of runoff in or out, as shown in Fig. 3. Moreover, to manage side overflow runoff, extracorrugated iron sheets were extended only for mechanical structures (SB and FJ). Each runoff plot was equipped with a trapezoidal runoff collection trench at the lower end (Figs. 3 and 4). This trench was lined with a 5-mm-thick geomembrane plastic sheet. Each collector trench had a capacity of 5.4 m^3, which was large enough to accommodate runoff from extreme rainfall events. In the experimental area, a manual rain gauge was set up to collect the daily rainfall amount. The total number of rainy days is defined as days with rainfall amounts greater than 0.1 mm, as reported by Bewket and Conway (2007).

Table 2
Treatments and descriptions.

LMP treatment	Description of treatment
Control	Farmers' Practice (conventional Practice): The plot was intensively tilled using a traditional ard plow, locally known as a *Maresha*, to a depth of 15−20 cm. Based on local experience, 95% of the crop residue was removed. The crop rotation included finger millet in 2021, soybeans in 2022, and maize in 2023.
Soil bund	The soil bund (SB) is an embankment constructed by digging a ditch along the contour and building an embankment below the ditch. The SB included a ditch 0.5 m deep and 0.5 m wide, with an embankment that had a compacted height of 0.5 m, a base width of 1.2 m, and a top width of 0.4 m. Intensive tillage was similar to the control treatment, removing over 95% of the crop residual. The crop rotation included finger millet in 2021, soybeans in 2022, and maize in 2023.
Fanya juu	Fanya juu (FJ), derived from the *Kiswahili* phrase meaning "throw the soil uphill", is similar to a soil bund, with the key difference being that the ditch is located below the embankment. The SB included a ditch that was 0.5 m deep and 0.5 m wide, with an embankment featuring a compacted height of 0.5 m, a base width of 1.2 m, and a top width of 0.4 m. Intensive tillage was done, removing over 95% of the residual material. The crop rotation practiced included finger millet in 2021, soybean in 2022, and maize in 2023.
Vetiver grass strip	Vetiver grass strips (VGS) (*Vetiveria zizanioides*) are deep-rooted perennial grass strips planted along the contour. The VGS dimensions include a strip width of 1 m, with a zigzag spacing of 15 cm between rows and plants. Intensive tillage was similar to the control treatment, removing over 95% of the crop residue. The crop rotation included finger millet in 2021, soybeans in 2022, and maize in 2023.
Conservation agriculture	In the current study, CA involved digging planting holes with no-tillage (direct seed sowing), maintaining 100% permanent crop residue cover throughout the year, manually eradicating weeds without the use of herbicides before seeding, and following a crop rotation sequence of finger millet in 2021, soybeans in 2022, and maize in 2023.

Fig. 3. A partial overview of runoff plots established for different land management practices (LMPs) on agricultural land in the study area.

2.4. Runoff and soil loss measurements

The runoff volume in each trench was measured every 24 h every morning at 8:00 a.m. Before representative runoff samples were collected, each runoff collector trench was mixed thoroughly to create turbulence. Then, 2-L runoff samples were taken using plastic bottles. One liter of sample was used to determine the sediment concentration (soil loss), and another liter was used for the determination of sediment-associated SOC and nutrient losses. The total runoff accumulated in each trench was measured via a 10-L plastic bucket and a 1-L graduated jug. These measurements were recorded for each rainfall event when runoff was generated and entered into trenches from runoff plots. A total of 307 runoff samples were collected during the study period: 169 samples in 2021, 63 in 2022, and 75 in 2023. Once the runoff data were recorded, each trench was then cleaned and emptied the next day.

Daily runoff samples were tag-leveled (date and treatment) and taken to the Pawe Agricultural Research Center for further analysis. The daily collected runoff samples were filtered through Whatman Grade 42 ashless filter paper with catalog numbers 1442−125. First, the empty filter paper was weighed via a digital balance (precision = 0.001 g). Then, the 1-L runoff sample was poured onto filter paper. The filter sediment was oven-dried at 105 °C for 24 h. After drying, the sediment was weighed again via a digital balance, and the weight of the filter paper was subtracted to determine the actual degree of soil loss. Finally, the sediment concentration was estimated from a 1-L runoff sample. The total soil loss was estimated by multiplying the daily sediment concentration (g/L) by the daily runoff volume (L) and summing over the year to determine the annual soil loss rate. To determine the actual runoff from each plot, direct rainfall into the runoff collection trench was subtracted from the total measured daily runoff (Fig. 4). The runoff depth (mm) was calculated by dividing the runoff volume (L) by the plot area (90.4 m^2). The runoff coefficient (RC) and runoff and soil loss conservation efficiency (RSCE) were calculated following the methods of Sahoo et al. (2016) and Sultan et al. (2018) as

$$\mathrm{RC}\ (\%) = \text{Daily runoff (mm)/Daily rainfall (mm)} \times 100 \quad (1)$$

$$\text{Soil conservation efficiency }(\%) = \left(\frac{\text{Soil loss from Control plots} - \text{Soil loss from LMP plots}}{\text{Soil loss from Control plots}}\right) \times 100 \quad (2)$$

$$\text{Runoff conservation efficiency }(\%) = \left(\frac{\text{Runoff from Control plots} - \text{Runoff from LMP plots}}{\text{Runoff from Control plots}}\right) \times 100 \quad (3)$$

The soil data were collected from two sources: the runoff plot experiment test site and the sediment collected from the trench. To evaluate the losses of sediment-associated SOC and nutrients from runoff plots, a similar procedure as that described for soil loss determination was followed. The only difference was that daily runoff samples were bulked into 30-L jars (with 5 treatments × 3 replications), and a total of 15 jars were prepared at the experimental site. The monthly bulk runoff samples were decanted from each jar, and the remaining sediment was packed into a plastic bag. These packed samples were then taken to the soil sample preparation room and air-dried. After the monthly sediment samples were collected for June, July, August, and September, the samples were carefully mixed, and approximately 300 g of the composite sediment was labeled and prepared for further analysis.

Soil samples were also collected from the experimental site during the establishment of the runoff plots to obtain baseline information at depths of 0−20 cm and 20−40 cm. The samples were taken from five sampling points and combined to create a composite sample. After the point samples were carefully mixed, approximately 1 kg of the composite sample was packed in a plastic bag and transported to the laboratory for further analysis. To determine the bulk density, undisturbed soil samples were

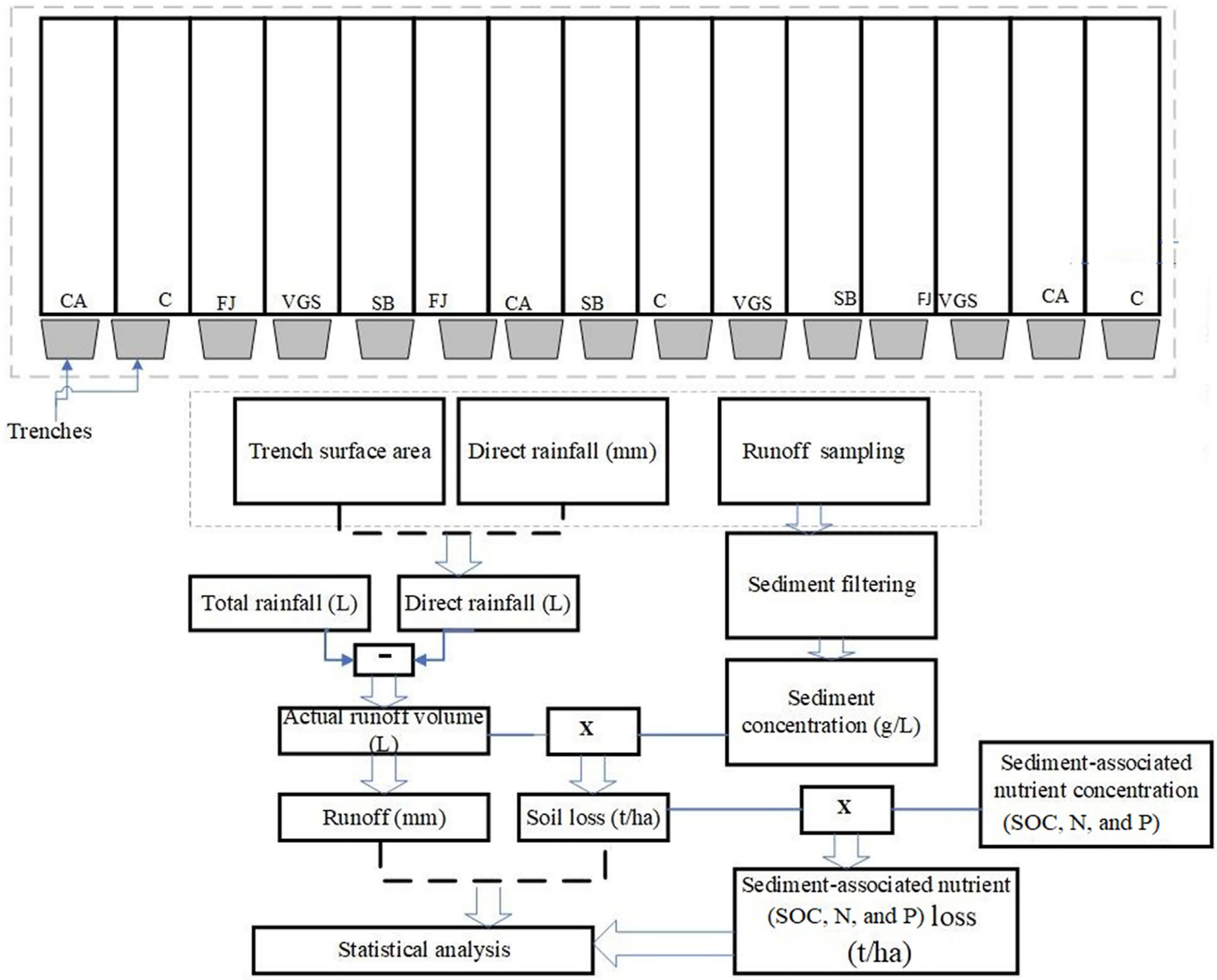

Fig. 4. General approach used to assess runoff, soil erosion, and sediment-associated SOC, N, and P from runoff plots involves collecting runoff samples and filtering sediments in the study area.

collected via a core sampler with a height and inner diameter of 5 cm each (approximate volume = 98.125 cm^3). The soil samples were air-dried in a soil preparation room, ground with a ceramic mortar and pestle, and finally sieved through a 0.5 mm sieve (Jones, 2001).

The soil particle size distribution was analyzed via the Bouyoucos (1962) hydrometer method. The soil bulk density was calculated via the core sampler method by dividing the oven-dry soil mass (dried at 105 °C for 24 h) by the core sampler volume (Wilke, 2005). The total porosity (TP) was determined via the method described by Weil and Brady (2017) Equation (4). The soil pH was measured potentiometrically at a 1:2.5 soil-to-water ratio (Van Reeuwijk, 2002). The SOC content was determined via the (Walkley & Black, 1934) wet digestion method. Total nitrogen was analyzed through the Kjeldahl digestion, distillation, and titration methods (Bremner & Mulvaney, 1982), whereas available P was measured in the extraction solution via the Bray II method (Bray & Kurtz, 1945). The cation exchange capacity (CEC) was measured via the ammonium acetate method at pH 7.0 after extracting the soil samples (Houba et al., 1989). The total loss of each nutrient, which was dependent on annual soil loss, was calculated by multiplying the total sediment remaining in each treatment by the average SOC and nutrient concentrations. The losses of SOC, N, and P associated with sediment were calculated as described by Lemma et al. (2017) as

$$\text{Soil porosity } (\%) = \left(1 - \frac{\text{Bulk Density}}{\text{Particle Density}}\right) \times 100 \tag{4}$$

$$\text{SOC loss (kg/ha)} = \text{SOC sediment associated concentrationin } \% \times 10 \times \text{Soil loss (kg/ha)} \tag{5}$$

$$\text{N loss (kg/ha)} = \text{N sediment associated concentration in } \% \times 10 \times \text{Soil loss (kg/ha)} \tag{6}$$

$$\text{P loss (kg/ha)} = \text{P sediment associated concentration in sediment (mg/kg)} \times 10^{-3} \times \text{soil loss (t/ha)} \tag{7}$$

where 10 is a conversion factor to convert N and C into g/kg and where 10^{-3} is the conversion of mg/kg P into g/kg.

2.5. Replacement cost estimation for sediment-associated nutrient losses

In this study, the cost of replacing sediment-associated nutrients lost due to soil erosion was estimated (Bojö, 1996; Mulualem et al., 2021). Commonly available fertilizers in the study area include urea (46N-0-0) and NPS (19N-38P_2O_5-0K-7S). The nutrient loss of N and P from each LMP, or the concentration of nutrients in the sediment (mg/kg), was converted to equivalent N and phosphorus pentoxide (P_2O_5) (or P) and then to urea and NPS fertilizers. First, the amount of P lost through sediment was translated to an equivalent amount of NPS. Next, the nitrogen content from the estimated NPS fertilizer was determined. The remaining nitrogen (the total N amount in the nutrient loss minus the N estimated from NPS) was then converted to an equivalent amount of urea fertilizer. Finally, the total amounts of NPS and urea fertilizers were converted to costs by multiplying the quantities by the respective fertilizer prices. The prices used to estimate the replacement cost of the exported nutrients in runoff were 4,342 Ethiopian Birr for 100 kg of NPS and 4,350 Ethiopian Birr for 100 kg of urea.

2.6. Statistical data analysis

Before the data were analyzed for normality, the Shapiro–Wilk, Bartlett, and Durbin–Watson tests (Shimizu et al., 2022) were applied. Analysis of variance (ANOVA) was then performed, followed by mean separation via the least significant difference (LSD) test ($p < 0.05$) for the runoff coefficient and sediment-associated SOC and nutrient loss, and the LMPs were compared with those of the control practices (Gomez & Gomez, 1984). Additionally, two-way ANOVA and Tukey's honest significant difference (HSD) test were used to assess runoff, soil loss, and the conservation efficiency of runoff and soil loss, with a significance level of $p < 0.05$ for treatment effects and year interactions (Gomez & Gomez, 1984). All the statistical analyses were performed via R software version 4.3.2.

3. Results and discussion

3.1. Effects of land management practices on runoff and soil loss

The effects of LMPs on runoff and soil loss were observed over three consecutive rainy seasons (Figs. 5 and 6). The annual runoff ranged from 57 to 463.2 mm in 2021, 49 to 375.4 mm in 2022, and 64.2 to 383.7 mm in 2023. These results were obtained from a total of 129 rainfall events (51 in 2021, 38 in 2022, and 40 in 2023) that generated runoff. Correspondingly, soil loss ranged from 3.8 to 47.4 t/ha in 2021, 3.7 to 41.8 t/ha in 2022, and 5.3 to 47 t/ha in 2023. Compared with the control plots, the field plots treated with FJ, SB, CA, and VGS significantly reduced runoff, which ranged from 49 to 69.5, 52.3 to 64.3, 83.5 to 232.5, and 200.5 to 367.3 mm, respectively, where the runoff depth ranged from 375.4 to 463.2 mm. The corresponding annual average soil loss for the field plots treated with FJ, SB, CA, and VGS ranged from 3.1 to 5.9, 3.7 to 5.3, 5.6 to 16.8, and 14.7 to 32.2 t/ha, respectively. All these values were significantly lower than those of the control plot, where soil loss ranged from 41.8 to 47.4 t/ha. In terms of the three-year averages, the runoff values were 58.5, 59.6, 136.9, and 260.3 mm greater for FJ, SB, CA, and VGS, respectively, than for the control plot (407.5 mm) (Fig. 3A in the ESM). The corresponding soil loss values were recorded as 4.3, 4.5, 11.1, and 21.5 t/ha, respectively, compared with the control plot (45.4 t/ha) (Fig. A4 in the ESM).

The ANOVA test results (Figs. 5 and 6 and Figs. A3 and A4 in the ESM) revealed that runoff and soil loss from the LMP-treated plots were significantly lower ($p < 0.05$) than those from the control plot during the experimental seasons. The highest runoff and soil loss values were recorded in the control plots, whereas the lowest values were recorded for FJ, followed by SB (Figs. 5 and 6). The runoff and soil loss values generally increased in the order of FJ < SB < CA < VGS. Considering the three-year averages, the mechanical measure soil loss rates were 4.3 t/ha for FJ and 4.5 t/ha for SB. These values were below the mean soil loss tolerance of 10 t/ha

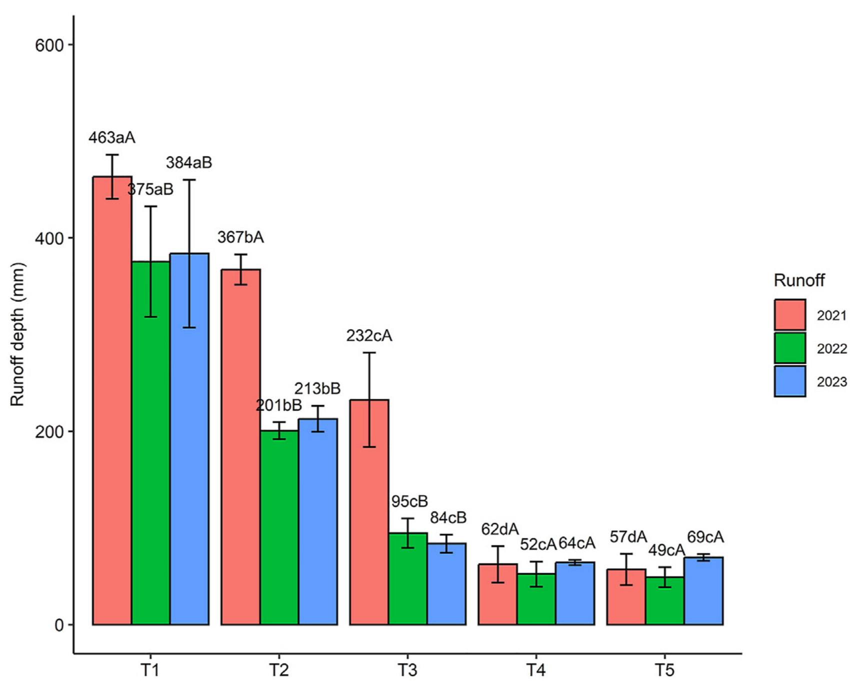

Fig. 5. Runoff rates under different land management practices in the Beles River basin from 2021 to 2023. Note: T1-control; T2-vetiver grass strip; T3-conservation agriculture; T4-soil bunds; and T5-Fanya juu. Means followed by the same lowercase letter between treatments and the same uppercase letter over the year do not differ significantly as determined by the Tukey test ($p < 0.05$).

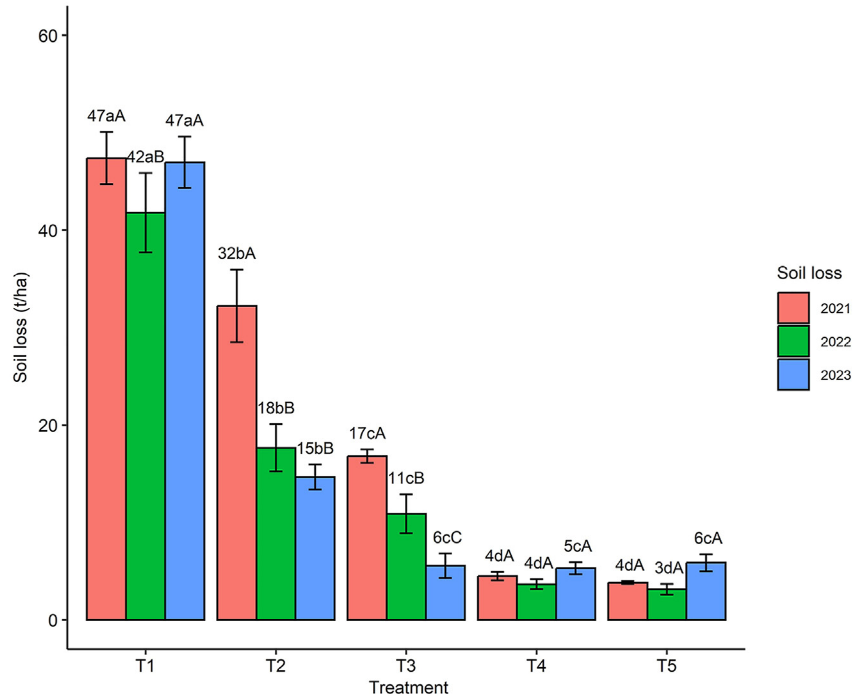

Fig. 6. Soil loss rates under different land management practices in the Beles River Basin from 2021 to 2023. Note: T1-control; T2-vetiver grass strip; T3-conservation agriculture; T4-soil bunds; and T5-Fanya juu. Means followed by the same lowercase letter between treatments and the same uppercase letter over the year do not differ significantly as determined by the Tukey test ($p < 0.05$).

suggested by Hurni (1983) for Ethiopian conditions. These values for mechanical measures are greater than those reported in other studies performed both in Ethiopia and elsewhere in Tanzania and Eritrea (Adimassu et al., 2014; Amare et al., 2014; Herweg & Ludi, 1999; Regasa & Nones, 2024; Sultan et al., 2018; Tenge et al., 2005; Wolka et al., 2018). This approach might be designed according to established technical standards and newly excavated structures that are robust enough to accommodate incoming runoff and soil loss. Moreover, this is likely associated with the intensive rainfall characteristics of the study areas. Previous authors (Belayneh et al., 2020; Mekonnen et al., 2016) also supported this finding, noting that newly excavated structures were more effective than older ones in reducing runoff and trapping sediment.

In addition to the measures for FJs and SB, CA practices significantly reduced runoff by 84–232 mm and soil loss by 8–16 t/ha over the years ($p < 0.05$) (Figs. 5 and 6). These findings clearly indicate that reductions in runoff and soil loss more than doubled between 2021 and 2023. This effectiveness is attributed to the synergistic effects of undisturbed (nontilled) soil and crop residues covering the surface, which protect the soil from direct raindrop impact, slow surface runoff, and promote water infiltration. Additionally, more than eight months of rainfall and high temperatures (Fig. 2) accelerated the decomposition rate, allowing crop residues to contribute additional organic matter throughout the year, thereby improving the soil structure and increasing infiltration (Araya et al., 2012, 2024; Ikazaki et al., 2018). Similarly, a study by Lanckriet et al. (2012) reported 14.4 t/ha of soil loss and 27.9 mm of runoff under CA practices in northern Ethiopia.

Vetiver grass strip practices such as CA also significantly reduced runoff and soil loss, varying from 201 to 367 mm and 15 to 30 t/ha, respectively, throughout the year ($p < 0.05$) (Figs. 5 and 6). Unlike mechanical measures, VGS biomass systems have developed gradually due to their nature. Their fast growth, increased tiller density, and vigorous biomass led to efficient runoff redistribution. This process dissipates runoff energy and reduces sediment concentrations at the plot outlet. Mekonnen et al. (2016) reported that VGS reduced soil loss by 29 t/ha and runoff by 69 mm in the highlands of Ethiopia, whereas Welle et al. (2006) reported that VGS reduced soil loss by 1.3 t/ha and runoff by 22 mm in the lowlands of Ethiopia. Moreover, Oshunsanya (2013) reported a reduction of 32 t/ha in soil loss elsewhere in Nigeria. Additionally, implementing grass strips is a proven measure to reduce runoff and soil loss through the progressive development of bench terracing (Kagabo et al., 2013; Kinoti & Gachene, 2015; Mekonnen et al., 2015). Similar observations have been reported by various other studies (Amare et al., 2014; Bu et al., 2008; Herweg & Ludi, 1999; Mengistu et al., 2016; Tu et al., 2018), indicating that grass strip measures were much more effective than physical terraces in reducing runoff and soil loss, particularly as they aged.

In addition to reducing runoff and soil loss, VGS significantly contributed to climate change mitigation through its biomass system. The current carbon stock contents are 6 kg C/km in 2021, 12 kg C/km in 2022, and 18 kg C/km in 2023 (the width, row spacing, and planting arrangement are described in detail in Table 2). These results clearly demonstrate that VGS doubled and tripled carbon sequestration during the study period (Fig. A5 in the ESM). Tessema et al. (2022) reported that vetiver grass biomass significantly increased soil carbon accumulation, with faster root decomposition accelerating its contribution to stable soil organic matter. Previous studies reported that vetiver grass has the potential to sequester carbon in Ethiopia and other countries, such as Australia (Tessema et al., 2020, 2022).

Generally, LMPs reduce runoff and soil loss by dissipating rainfall energy or slowing runoff velocity, thereby redistributing surface water, promoting accumulation, and forming cross-slope barriers. The current findings are in line with those of Belayneh et al. (2020), Ebabu et al. (2019), and Sultan et al. (2018), demonstrating that LMPs achieved more effective reductions in the upper Blue Nile

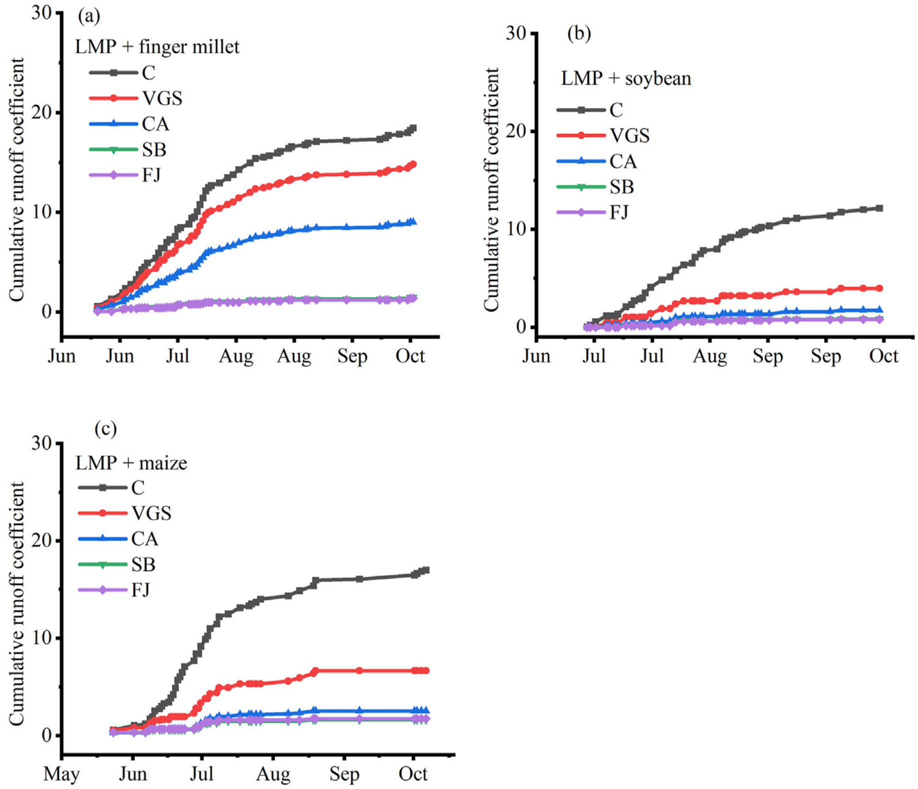

Fig. 7. Relationships between the cumulative runoff coefficient under different LMPs with varying crop covers during the rainy seasons of the study. Note: C-control, VGS-vetiver grass strip, CA-conservation agriculture, SB-soil bund, and FJ-fanya juu: (a) 2021; (b) 2022; and (c) 2023.

River Basin. A nationwide review by Desta et al. (2021b) revealed that mechanical measures, such as SB and FJ, were more effective than were agronomic practices (e.g., mulching and intercropping) and soil management techniques (e.g., conservation tillage) in reducing soil loss on cropland, with soil loss ratios ranging from 0.08 to 0.09, 0.08 to 0.27, and 0.38 to 0.67, respectively, across Ethiopia. Similarly, Wolka et al. (2018) reported comparable effects in East Africa, with LMPs reducing runoff and soil loss by from 13% to 71%, 39% to 83%, and 25% to 60%, respectively. Desta et al. (2021a) also reported soil loss reductions ranging from 0.5 to 55 t/ha under LMP field conditions. These variations in runoff and soil loss may be attributed to environmental factors such as crop cover, soil properties, rainfall, and topography, as confirmed by Desta et al. (2021a) and Wolka et al. (2018), which influence the effectiveness of LMPs.

Figs. 5 and 6 resulted in greater runoff and soil loss in the first year across all plots, likely because of the high rainfall (Fig. A1 in the ESM) associated with freshly prepared runoff plots. Despite significant rainfall differences among the first, second, and third years, runoff and soil loss remained similar, possibly because of the effects of cover crops under the millet–soybean–maize rotation system. This variation may result from both rainfall variability and the influence of cover crops. Molla (2024) and Molla and Desta (2022) reported that, compared with maize, soybean reduced runoff and soil loss by 43.3% and 43.4%, respectively, and that, compared with maize, soybean reduced runoff by 16.6% and soil loss by 39.4%, respectively, at the same experimental site and in the same season. Desta et al. (2021a) reported that maize was one of the least effective cover crops for reducing soil loss. Adimassu and Haile (2011) reported runoff amounts of 144.7, 181.4, and 169.5 mm in field pea, faba bean, and wheat plots, respectively, with corresponding soil losses of 16.9, 29.9, and 20.29 t/ha, respectively. With respect to the effects of the plant population, closely planted maize reduced runoff by 47 mm and soil loss by 3.9 t/ha, whereas wider-planted maize resulted in 51 mm of runoff and 7.2 t/ha of soil loss (Mohammed & Gumbs, 1982). These results demonstrate that runoff and soil loss are also influenced by both the type of cover crop and the crop management practices.

3.2. *Efficiency of land management practices*

Land management practices are effective not only because of their ability to reduce the rates of soil loss or runoff but also because of their ability to reduce runoff and soil loss over the years. The efficiency of LMP in reducing runoff and soil loss was evaluated via the runoff coefficient (RC) and soil and runoff conservation efficiency (SRCE), as shown in Fig. 7 and Figs. A6 and A7 in the ESM. This RC knowledge is a prerequisite for designing effective runoff-control measures, estimating water yield in the basin, and forecasting flood risks. The lowest RC values were observed in the SB- and FJ-treated fields, followed by those in the CA, VGS, and control fields, with mean values of 5.7%, 12.7%, 24.2%, and 38.3%, respectively (Fig. A7 in the ESM), which were lower than those in the control. The RC is influenced by many factors (soil type, land use, topography, vegetation, and climatic conditions) (Liu et al., 2020). However, this variability in the RC was attributed primarily to rainfall interception and retention by the LMPs (Fig. 7 and Fig. A7 in the ESM).

A graphical plot illustrates the cumulative RC on the y-axis and the daily rainfall on the *x*-axis (Fig. 7). Each point represents the RC observed after a rainfall event that generated the runoff. The graphs display data from all five treatment plots (Fig. 7). High cumulative runoff coefficients were observed for the control and VGS plots, which implies a lower effective runoff retention capacity of the practices but a lower cumulative RC for FJ and SB, reflecting better performance in retention runoff.

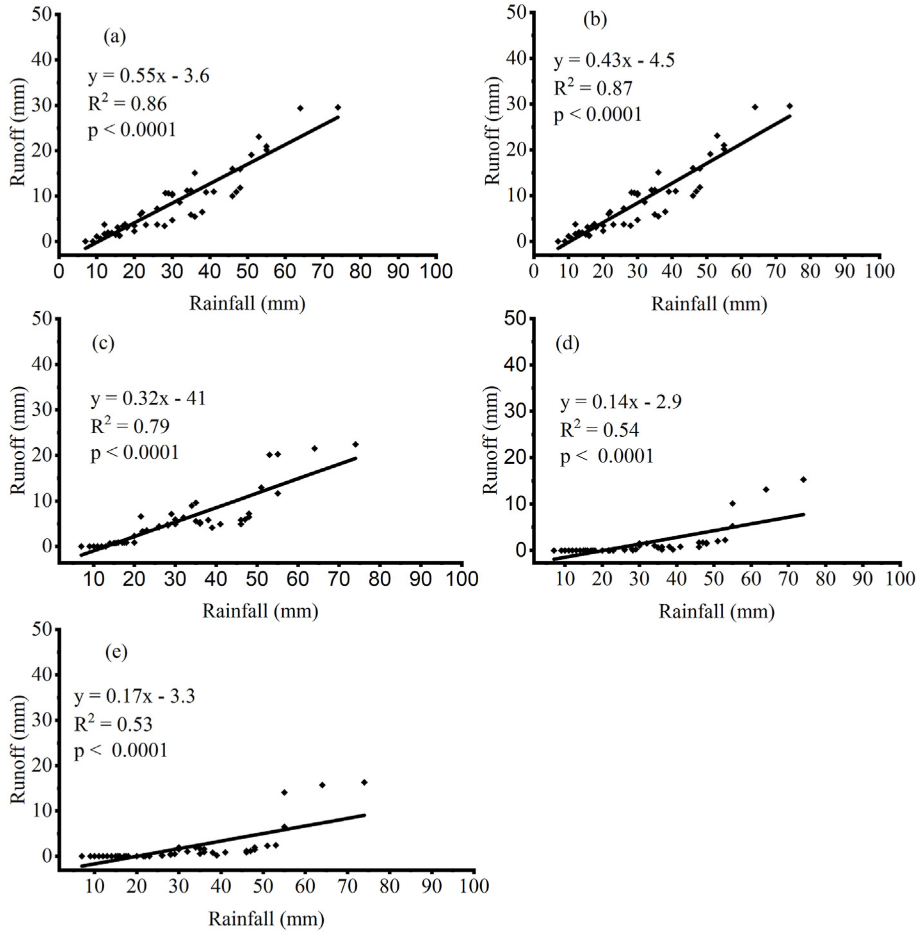

Fig. 8. Regression curves fitted to daily rainfall and runoff data under different LMPs in the study period. (a) Control, (b) vendor grass strips, (c) conservation agriculture, (d) soil bunds, and (e) Fanya juu. Note: R^2 is the coefficient of determination.

The cumulative RCs in the CA and VGS plots remained between those in the control plot and those in the FJ and SB plots and gradually decreased over time. This clearly shows that CA and VGS improved runoff retention over time, approaching the performance of FJ and SB (Fig. 7). Compared with the other LMPs and the control, the mechanical measures (FJ and SB) had more or less similar cumulative RC retentions of runoff and were more efficient. The control plot line diverges from the LMPs, clearly demonstrating an increase in runoff over the year (Fig. 7). These findings are consistent with those of previous studies (e.g., Sultan et al., 2018) reporting that soil and water conservation practices could reduce the RC in the Upper Blue Nile River Basin. Another study revealed that the RC rates increased in the following order: 31.1% for newly constructed soil bunds, 41.8% for soil bunds aged 11 years, and 47.8% for the control (Belayneh et al., 2020). Previous studies also reported that low RCs were observed for areas treated with mechanical structures (Adimassu et al., 2014; Lemma et al., 2018; Taye et al., 2013). Therefore, RCs depend on the types of land management measures and their duration. Understanding RCs is pivotal for the effective management of hydrology and water resources.

Over the past three years, in addition to LMPs, crop covers such as finger millet, soybean, and maize have contributed to variations in RC. As shown in Figs. 7(a) and 7(c), both the finger millet and the maize crop cover experienced a sharp increase in the RC until the end of July, followed by a gradual increase until early September. The soybean (Fig. 7(b)) cover crop showed a small rate of change in the RC from the beginning to the end of the season, suggesting more effective runoff retention than did maize and finger millet. Wang et al. (2021) noted that soybeans were more effective than maize in reducing RC. Molla (2024) reported significant reductions in RC with soybean via conventional tillage (40.5%) compared with maize via conventional tillage (59.8%) in the lowlands of Ethiopia. However, the cumulative RC results show that combining cover crops with mechanical structures was more effective in reducing runoff and improving the retention and storage of excess runoff. Various authors have reported similar observations under different annual cover crops (Adimassu et al., 2020; Adimassu & Haile, 2011; Kebede et al., 2021). Overall, the cumulative RC results of the current study show that combining cover crops with mechanical structures is more effective in reducing runoff and improving the retention and storage of excess runoff.

The runoff conservation efficiency for FJ, SB, CA, and VGS ranged from 81% to 88%, 83% to 87%, 50% to 78%, and 21% to 46%, respectively (Fig. A6 in the ESM). Similarly, the soil loss conservation efficiency for these measures ranged from 87.5% to 92.1%, 88.7% to 91.3%, 64.3% to 83.9%, and 31.5% to 68.2%, respectively. The ANOVA results indicated highly significant differences ($p < 0.05$) in the effects of LMPs on the RSCE. The mean annual RSCE was highest for the mechanical measures (FJ and SB), followed by CA and VGS, with values of 85.3% and 90.7% for FJ, 85.2% and 90.1% for SB, 67.6% and 74.1% for CA, and 36.3% and 52.4% for VGS, respectively. The mechanical measures (FJ and SB) did not significantly vary ($p < 0.05$) in terms of the RSCE throughout the year (Fig. 8), whereas CA and VGS significantly varied ($p < 0.05$) in terms of the RSCE throughout the

Table 3
Results of the analysis of variance (ANOVA) for sediment-associated SOC, N, and P losses under different land management practices between 2021 and 2023.

LMP	Soil organic carbon (kg/ha)				Total nitrogen (kg/ha)				Available phosphorus (kg/ha)			
	2021	2022	2023	Mean	2021	2022	2023	Mean	2021	2022	2023	Mean
C	1,268.3^{a}	1,010.8^{a}	1.190.1^{a}	1,156.4^{a}	86.3^{a}	70.2^{a}	105.8^{a}	87.4^{a}	0.53^{a}	0.47^{a}	0.37^{a}	0.46^{a}
VGS	826.2^{b}	393.2^{b}	356.9^{b}	525.4$_{b}$	64.9^{b}	28.2^{b}	31.9b	41.6^{b}	0.43^{b}	0.27^{b}	0.3a^{b}	0.33^{b}
CA	116.93^{c}	208.7^{c}	148.4^{c}	267.4^{c}	32.5c	14.8^{c}	12.5^{c}	19.9^{c}	0.3^{c}	0.2a^{b}	0.23b^{c}	0.24^{c}
SB	445^{d}	91^{c}	133.3c	113.7^{d}	8.4^{d}	6.4^{c}	11.6^{c}	8.8^{d}	0.13^{d}	0.1^{c}	0.01^{d}	0.11^{d}
FJ	99.5^{d}	81.3^{c}	161.1^{c}	113.9^{d}	6.6^{d}	5.2^{c}	15.1^{c}	8.9^{d}	0.1^{d}	0.1^{c}	0.2^{c}	0.13^{d}
CV	7.3	25.6	20.9	23.5	8.1	24.9	20.3	29.8	14.9	24.16	19.39	22.92
Sig	***	***	***	***	***	***	***	***	***	***	***	***
LSD	75.3	172.2	156.31	98.28	6.09	11.72	13.53	9.53	0.08	0.1	0.08	0.05

Note: a,b, c, and d: means followed by the same column letters between treatments in a column do not differ significantly as determined by the LSD test ($p < 0.05$) and *** = highly significant different at $p \leq 0.001$.

year. This result shows that the FJ and SB measures demonstrated maximum efficiency in conserving soil and water, whereas the CA and VGS measures gradually improved their efficiency in terms of soil and water conservation (Fig. A6 in the ESM). The reason is that newly constructed FJs and SB breakdown surface runoff velocity and have accumulation capacity for sediment storage in ditches and embankments.

CA practices synergistically protect soil and water by dissipating raindrop impact through surface residue cover while minimizing soil disturbances, enhancing soil structure, increasing porosity, and promoting infiltration. Similarly, VGS improved the RSCE throughout the year because of its rapid lateral growth, abundance of tillers, and high tiller density, effectively preventing soil erosion by water. In line with these results, Sultan et al. (2018) and Ebabu et al. (2019) reported runoff conservation efficiencies ranging from 25% to 73% from land management practices in the Upper Blue Nile River Basin. Similar studies in northern Ethiopia reported conservation efficiencies of mechanical measures ranging from 29% to 72% for runoff and from 41% to 86% for soil loss (Ebabu et al., 2023). Similarly, in India, Sahoo et al. (2016) reported that, compared with vegetative barrier measures, soil and water conservation measures resulted in decreases of 44.5% and 46.5%, respectively, in soil loss and runoff conservation.

The current findings on the soil loss conservation efficiency of CA practices revealed a rate of 74%, which is comparable to the 59% reported by Lanckriet et al. (2012) in northern Ethiopia and the 63% reported by Kebede et al. (2023) in the central highlands of Ethiopia. Similarly, the soil and water conservation efficiencies of the VGS were 31.5% and 68.2%, respectively. Consistent with this, Mekonnen et al. (2016) reported that grass strips demonstrated a sediment trapping efficiency of 58% for vetiver grass in the Upper Blue Nile River basin (Welle et al., 2006) and reported a VGS trapping efficiency of 59% in the lowlands of Ethiopia, with an annual mean rainfall of 661 mm. The current results generally indicate that soil conservation efficiency is greater than water conservation efficiency.

The efficiency of the LMP treatments was also evaluated via regression curves fitted to daily rainfall and runoff data from both the treated and control plots (Fig. 7). The control and VGS plots presented the strongest correlations with rainfall (0.95 and 0.94, respectively), whereas SB and FJ presented weaker correlations (0.69 each) (Table A1 in the ESM). Mechanical measures (FJ and SB), with their lower slope coefficients (0.14 and 0.17, respectively), appeared to be the most efficient at reducing runoff, making them ideal for managing water in areas prone to erosion. The regression curves (Fig. 8) and runoff parameters (Table A1 in the ESM) indicated statistically significant correlations in both the LMP and control plots during the study period, with R^2 values ranging from 0.53 to 0.87 ($p < 0.0001$). The slope coefficients of the regression curves reflect the efficiency of runoff under different LMP conditions. A higher slope suggests a stronger relationship between rainfall and runoff, indicating lower runoff conservation efficiency (as seen for the control and VGS). In contrast, lower slopes, as observed for FJ and SB, signify greater efficiency in reducing runoff, making these methods more effective for water management in erosion-prone areas. Overall, the runoff conservation efficiency was ranked in ascending order as FJ, SB, C, and VGS, with corresponding regression slope values of 0.14, 0.17, 0.32, and 0.45, respectively. This result clearly showed that the effectiveness of LMPs resulted in reduced runoff in storage embankments and channels as well as significant sediment trapping (Ebabu et al., 2019, 2023; Sultan et al., 2018). In the control plot, a steep regression slope, high coefficient of determination, and strong correlation indicate how quickly runoff depth increases with increasing rainfall in the study area. Therefore, site-specific and agroecological studies are essential for generating model input parameters for runoff and soil erosion prediction, as well as for informing decision-making processes.

3.3. Effects of LMP on soil organic carbon and nutrient losses

Table 3 presents the results of the sediment-laden nutrient losses. In 2021, the maximum losses of sediment-associated SOC, N, and P were notably observed in both the treated and control plots.

Table 4
Mean nutrient loss and replacement costs of urea and NPS fertilizers in the current study.

LMP	Sediment associated nutrient loss (kg/ha)		Urea loss		NPS loss		Total cost
	N	P	(kg/ha)	Cost (ETB/ha)	(kg/ha)	Cost (ETB/ha)	ETB/ha (USD/ha)
C	87.44	0.46	188.9	8,217.2	2.8	121.6	8,338.7 (146.7)
VGS	41.64	0.33	89.7	3,901.9	2.0	86.8	3,988.7 (70.1)
CA	19.97	0.24	42.8	1,861.8	1.4	60.8	1,922.6 (33.8)
SB	8.79	0.11	18.8	817.8	0.7	30.4	848.2 (14.9)
FJ	8.97	0.13	19.2	835.2	0.8	34.7	869.10 (15.3)

Note: ETB = Ethiopian Birr.

The annual average values for SOC loss ranged from 113 to 1,190 kg/ha, N loss ranged from 8.8 to 87 kg/ha, and P loss ranged from 0.11 to 0.46 kg/ha during the study period. The ANOVA results revealed that sediment-associated SOC and nutrient losses were significantly lower ($p < 0.05$) in the LMP plots than in the control plots. The highest mean annual losses of SOC, N, and P were recorded in the control plot at 1,156.4, 87.4, and 0.46 kg/ha, respectively, whereas the lowest sediment-associated SOC and nutrient losses were observed with mechanical practices (FJ and SB).

Since sediment-associated SOC and nutrient losses depend on annual total soil loss, improved land management practices resulted in the lowest nutrient losses. Over the three-year period, there were no significant differences in sediment-associated SOC or nutrient loss between the FJ and SB treatments. However, the CA and VGS measures also substantially reduced sediment-associated SOC and nutrient losses compared with those in the control. The lowest nutrient loss associated with CA and VGS demonstrated the filtering and retaining capacity of nutrient-laden sediment before it was removed from the field/plot boundaries. The significant removal of SOC and N from the control plots was due mainly to accelerated soil erosion in the absence of erosion control barriers. Intensive tillage, which breaks down soil aggregates, further exposes soil to erosion.

Nitrogen, primarily found as nitrate (NO_3^-) and ammonium (NH_4^+), is highly mobile and soluble in water (McLatchey & Reddy, 1998). Compared with nitrogen, phosphorus, in the form of phosphate (PO_4^{3-}), has lower mobility and solubility in soil (Weil & Brady, 2017). The N and P losses from the control plots were greater than those reported by Haregeweyn et al. (2008) in northern Ethiopia but lower than the N and P losses of 154.7 and 1.84 kg/ha, respectively, reported by Selassie and Belay (2013) in northwestern Ethiopia. However, the current findings in central Ethiopia, as reported by Adimassu et al. (2014), show similar seasonal losses of SOC (930 kg/ha) and P (0.59 kg/ha) due to erosion from conventionally cultivated lands. Similar erosion-related sediment-associated losses of SOC, N, and P for conventional land use have also been reported in Kenya (Okeyo et al., 2014).

Over three years, the LMP-treated plots presented reductions in SOC, N, and P losses. Compared with those of the control, SOC losses were reduced by 90%, 90%, 77%, and 55% for FJ, SB, CA, and VGS, respectively. N losses were reduced by 90%, 70%, and 52%, whereas P losses were reduced by 72%, 76%, 28%, and 28%, respectively. In line with these results, Wolka et al. (2018) reported that soil bunds and Fanya Juu presented greater reductions in SOC and nutrient losses because of their ability to accumulate sediment in ditches or embankments. The current findings for SOC and nutrient loss reduction by FJ and SB (90%) were lower than the 96% reported by Wolka et al. (2021) for southern Ethiopia but much higher than the 65% reported by Mulualem et al. (2021) for northwest Ethiopia. In Kenya, a 95.5% reduction in SOC and nutrient losses was reported by Nathan et al. (2022), and in India Mahajan et al. (2021), significant reductions in SOC and nutrient losses resulting from land management practices were reported by.

As listed in Table 3, the highest SOC was removed from the control (cultivated land) (1,011–1,268 kg/ha), whereas the lowest SOC yield (114–525 kg/ha) occurred from fields managed by erosion control techniques. The reason was that conventional tillage disrupts soil aggregation and exposes soil to raindrops, leading to the removal and redistribution of SOC (Lal, 2014). These results clearly demonstrate that LMPs enhance soil carbon stocks, improve soil health, and simultaneously mitigate climate change. The current findings show that the P content in the sediment (0.46 kg/ha) was much greater than the P losses reported for land use in the Upper Blue Nile River basin (0.2 kg/ha) by Mulualem et al. (2021) and Girmay et al. (2009) in northern Ethiopia. In contrast, Adimassu et al. (2020) reported an even higher P content in sediment (1.3 kg/ha) from the central highlands of Ethiopia. The current results clearly demonstrate that the P content in sediment is generally lower than the values reported in other studies, which might be due to the inherent properties of the soil and P fixation due to soil acidity at pH 5.4.

3.4. Replacement cost of soil nutrient losses by erosion

To estimate the replacement cost of nutrient losses caused by erosion, the amounts of N and P lost were converted to the equivalent amounts of urea and NPS commercial fertilizer and then multiplied by the current fertilizer prices. The chemical fertilizers used in the study area are urea (46N-0P-0K) and NPS (19N-38P_2O_5-7S), which are the only fertilizers imported and distributed locally. In 2023, the average price of fertilizer was 4,350 Ethiopian Birr for 100 kg of urea and 4,342 Birr for 100 kg of NPS (exchange rate: 1 USD = 56.9 Birr). Table 4 lists the urea and NPS needed to replenish the N and P losses, as well as the total cost of fertilizers needed to replace the nutrients lost to erosion. Without any LMPs, approximately 8,338.7 Birr/ha/y would be required to purchase fertilizers to replenish the lost N and P induced by erosion. This highlights the importance of LMPs as a strategic approach to minimize degradation and fertilizer investment costs.

The results revealed a variation in replacement costs among the different LMPs, with the highest costs for the control plot and the lowest for the mechanical measures (SB and FJ) (Table 4). The annual fertilizer replacement costs were lowest for SB (848.2 Birr/ha), followed by FJ (869.9 Birr/ha), CA (1,922.6 Birr/ha), and VGS (3,988.7 Birr/ha). Although both nutrients are essential limiting factors for plant growth, nitrogen has the highest replacement cost for lost nutrients. Therefore, land management practices constitute a key strategy for minimizing the replacement cost of soil nutrients lost due to erosion. Mulualem et al. (2021) reported that nutrient losses due to water erosion cost averages of 1,387.6, 1,221.1, and 946.3 Birr/ha for the Guder, Aba Gerima, and Dibatie locations, respectively. Therefore, effective LMPs not only improve soil productivity and soil health but also help farmers reduce their production costs. In Europe, Panagos et al. (2018) reported that soil erosion by water is the major cause of soil nutrient depletion and economic loss in agroecosystems. Therefore, implementing erosion control techniques is crucial for enhancing and maintaining land ecosystem functions and services.

4. Conclusions

The current study examined the effects of various LMPs on runoff, soil, and sediment-associated SOC and nutrient (N and P) losses in the rainfed agroecosystem of the Beles River Basin in the subhumid lowlands of Ethiopia. The mean annual runoff, soil, and sediment-associated losses of SOC, N, and P from agricultural land (control plot) on a 9% slope were measured at 407.5 mm, 45.4 t/ha, 1,156 kg SOC/ha, 87.4 kg N/ha, and 0.46 kg P/ha, respectively. The results revealed that runoff, soil loss, and sediment-associated SOC and nutrient losses were significantly lower ($p < 0.05$) in the plots treated with LMPs than in the control plots. Mechanical measures (FJ and SB) were the most effective and best at reducing soil loss to 4.3 and 4.5 t/ha, with soil conservation efficiencies of 90.7% and 90.1%, respectively. These soil loss rates were lower than the mean soil loss tolerance (10 t/ha) under Ethiopian conditions. The soil and water conservation efficiencies of the FJ and SB measures continued to be consistent over the years, whereas the CA and VGS measures significantly increased the soil conservation efficiency by 66% and 54%, respectively, over the study period. The high efficiency achieved by CA over the years was due to the lack of mechanical soil disturbance, permanent soil cover, and crop rotation components, which dissipated raindrop impact and improved soil aggregation, leading to increased water infiltration. Similarly, the higher efficiency of VGS was attributed to their faster growth rate and denser root structure, which helped trap soil and reduce runoff. In addition, VGS offers the added benefit of sequestering carbon through its biomass system, significantly contributing to climate change mitigation efforts.

The effectiveness of LMPs in reducing SOC loss was most pronounced during the main rainy season, which produced the highest erosion rates in the study areas. The SOC loss in LMP runoff plots decreased by 113.9–525.4 kg/ha, underscoring the importance of implementing LMPs in subhumid lowland agroecological conditions for mitigating carbon losses and enhancing adaptation strategies. Establishing LMP also led to a decrease in nutrient loss of 9–42 kg/ha, resulting in reduced replacement costs for fertilizer inputs (urea and NPS), with savings ranging from \$4.9 to \$70.1 ha/y. These practices not only lower input costs for farmers but also improve soil health, as synthetic fertilizers can lead to emissions and harm beneficial soil microorganisms (Feliciano et al., 2022).

Understanding the potential of various land management measures (mechanical, biological, and agronomic) is crucial for end users. The current study provides valuable information for farmers and land managers in the middle Beles River Basin, supporting the planning and implementation of LMPs and appropriate land use in similar climatic and agroecological settings. The current findings can also guide experts and planners in developing land and water conservation strategies via a watershed approach, thereby advancing the knowledge base of contextualized land management practices in lowland agroecosystems.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

CRediT authorship contribution statement

Yenesew Assaye: Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Gizaw Desta:** Writing – review & editing, Visualization, Validation, Supervision, Software, Methodology, Investigation, Formal analysis, Conceptualization. **Eyayu Molla:** Writing – review & editing, Visualization, Validation, Supervision, Methodology, Formal analysis, Data curation, Conceptualization. **Zenebe Adimassu:** Writing – review & editing, Software, Methodology, Formal analysis.

Data availability statement

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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