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Effects of land uses on runoff, soil, and nutrient losses in the middle Beles River Basin, Northwest Ethiopia

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ABSTRACT

In 1984/85, the resettlement program of Ethiopia relocated farmers to the Beles River Basin, but unsustainable agricultural practices have since led to declining productivity and sedimentation of reservoir. To guide sustainable land management (SLM), this study assessed the impact of three land uses - cultivated land (CL), grazing land (GL), and zero grazing land (ZG) on runoff, soil and nutrient losses and carbon stock. Using a randomized complete block design with three replications, data were collected for two years from runoff plots on rainfall, runoff, soil loss, nutrient loss, carbon stocks, and soil moisture content. Results show significant differences ($P < 0.05$) among land uses. GL generated the highest runoff (493 mm), followed by CL (424 mm) and ZG (234 mm). Soil loss in CL generated (44 t ha⁻¹) which as four times higher than GL (11 t ha⁻¹) and eighteen times higher than ZG (2.4 t ha⁻¹). CL also recorded the highest losses of organic carbon (1139 and 78 kg ha⁻¹), nitrogen (78 kg ha⁻¹). Similarly, the highest available phosphorus (P) loss was recorded in CL (0.9 kg ha⁻¹). Critically, soil erosion rate in CL exceeded the country's permissible limit of 10 t ha⁻¹. This implies that severe soil erosion risk in CL and underscore the urgent need for implementing and adopting SLM practices to restore and enhance agricultural productivity and reduce off-site sedimentation and overall degradation of ecosystems in CL areas.

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Beles River Basin; land degradation; Ethiopia; land use; sustainable land management

1. Introduction

Soil erosion represents a significant contributor to land degradation on a global scale, with human-induced activities identified as the primary driver (Lal 2014). The process of deforestation, overgrazing, poor agricultural practices, and urbanization has all contributed to a substantial increase in soil erosion (Meshesha *et al.* 2024). In sub-Saharan Africa (SSA), soil erosion represents a significant impediment to sustainable agricultural development and food security, resulting in the loss of approximately 280 million tons (t) of grain annually (Wolka *et al.* 2018). Consequently, it is associated with an economic impact of approximately US\$9.3 billion yr⁻¹ (Chinzila 2018). Ethiopia, like other countries in SSA, is severely affected by soil erosion, with an estimated 1.9 billion t of soil lost annually (Fenta *et al.* 2021). The extent of soil loss varies considerably, with figures ranging from 1 t ha⁻¹ (Dizi) to more than 212 t ha⁻¹ yr⁻¹ (Andit Tid) (Herweg and Stillhardt 1999), while the average soil loss in Ethiopia ranges from 50 to 140 t ha⁻¹ (Desta *et al.* 2021). In comparison to other land uses, the highest soil loss was observed in CL, with

mean rates between 50 and 140 t ha⁻¹ and a maximum of 220 t ha⁻¹ (Desta *et al.* 2021), followed by grazing land (32.9 t ha⁻¹) (Adimassu *et al.* 2020, Jiang *et al.* 2020, Kindu *et al.* 2015) in Ethiopia. Therefore, the soil loss rates exceeded the tolerable limit (2–18 t ha⁻¹), as reported by Hurni (1983) for Ethiopian conditions. This indicates the necessity for immediate action to address the issue of soil erosion and rehabilitate degraded areas, with the aim of enhancing the overall system of productivity.

The associated soil erosion and other factors triggering the nutrient depletion rates at the national level were estimated to be 3700, 122, 13, and 82 kg ha⁻¹ yr⁻¹ for SOC, N, P, and K, respectively (Haileslassie *et al.* 2005, Van Beek *et al.* 2019). The annual cost of soil erosion and other related factors in Ethiopia is estimated at received an annual cost of \$4.3 billion (Gebreselassie *et al.* 2016). Consequently, Ethiopian agricultural productivity is generally substantially below global and regional standards, as reported by Tamene *et al.* (2017).

In order to address the issue of soil erosion, the Government of Ethiopia has implemented land

management practices across a range of land uses since the 1970s, particularly in areas prone to drought (Herweg and Ludi 1999). This has been done in collaboration with various stakeholders. These efforts continued until replaced by a watershed-based approach in the 1980s (Haregeweyn *et al.* 2015). However, soil erosion persisted, and in order to reinforce these efforts, the nationwide and holistic sustainable land management program was initiated in 2008 (Schmidt and Tadesse 2017). Consequently, 7.7 million ha (23%) of the Ethiopian highlands have subjected by land management measures (Bantider *et al.* 2019). These interventions have brought the benefits to land users, including enhanced soil fertility (Sinore *et al.* 2018), increased crop yield (Adgo *et al.* 2013), improved fodder productivity (Baudron *et al.* 2015), and enhanced production of both wood and non-wood forest products with enriched biomass and carbon stocks (Biratu *et al.* 2021). Moreover, exclosures significantly enhanced soil quality and health and provided economic returns (Nigussie *et al.* 2020). It is essential to restore degraded land through the implementation of promising technologies that can support livelihoods and provide ecosystem functions and services.

The Upper Blue Nile Basin experiences the highest degree of soil erosion in Ethiopia, with an average loss of 46 t ha^{-1} and a total annual loss of 573 million t (Ali *et al.* 2014, Haregeweyn *et al.* 2017, Fenta *et al.* 2021). 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 $0.62 \text{ million t y}^{-1}$ 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.

In studies, runoff, soil erosion, SOC, and nutrient losses have investigated under different land uses in Ethiopia's highlands (Herweg and Ludi 1999, Adimassu *et al.* 2020, Wolka *et al.* 2021, Ebabu *et al.* 2023). However, their potential impacts depend on specific agroecological settings and local conditions (Abera *et al.* 2020, Desta *et al.* 2021, Eshetie *et al.* 2024). Nevertheless, research in relation to these aspects has been scarce in Ethiopia's lowlands in general and the Beles River Basin in particular. 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 in comparison to highlands. There is a lack of information available on mini-watersheds (Teklemariam *et al.* 2017), and some studies focus on physical models (Kebede *et al.* 2021) without sufficient experimental validation. Furthermore, Nyssen *et al.* (2018) highlighted the

necessity for further research on soil erosion, which they identified as a significant issue affecting land use dynamics in the Beles River Basin and requires further research. The rapid conversion of natural forests into agroecosystems, driven by mechanized agriculture and large resettlement programs without adequate mitigation measures, may collectively contribute to the siltation of the GERD (Teklemariam *et al.* 2017). Since the basin is directly connected to the GERD, siltation is a serious concern. Thus, it is essential to examine the current status of land use for land management practitioners, experts, and other stakeholders. Therefore, the objective of this study was to evaluate runoff, soil loss, and sediment-associated nutrient and SOC losses under different land uses in the Beles River Basin.

2. Materials and methods

2.1. Study area description

The Tana-Beles Resettlement Villagization is located in the Middle Beles River Basin, Upper Blue Nile Basin, western Ethiopian escarpment. It is situated between 11.2° – 11.35° N and 36.33° – 36.53° E, covering a total area of 220,000 ha (Zanardi 2011). The study area is located 177 km east of the GERD and 572 km northwest of Addis Ababa, the capital city of Ethiopia. It is also situated between Lake Tana and directly connected to the GERD reservoir (Figure 1). The Tana-Beles Resettlement Villagization area is one of the largest resettlement sites in the country, covering an area of 220,000 ha with about 60 km length and 30 km width its altitude varies between 710 and 1300 m.a.s.l (Abbute 1997). The geology of the middle Beles River Basin 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). Based on the data obtained from the Tana-Beles Project Office, the soils of the valley are broadly categorized as 47% Nitisols (red soils), which are easily worked though not very fertile, and 53% Vertisols (black soils) (Abbute 1997).

The long-term (1987–2022) mean annual minimum and maximum temperatures were recorded at 16.6 and 32.7°C , respectively, as shown in Figure 2. Maximum temperatures can reach as high as 42°C during the period from mid-February through May, which are typically the hottest months. In contrast, the coldest months are December and January. The long-term annual mean rainfall, about 1600 mm, shows a unimodal pattern extending from April through November (Figure 2). Rainfall exceeds evapotranspiration from June to October, while in the other months, evapotranspiration rates are higher (Figure 2). The

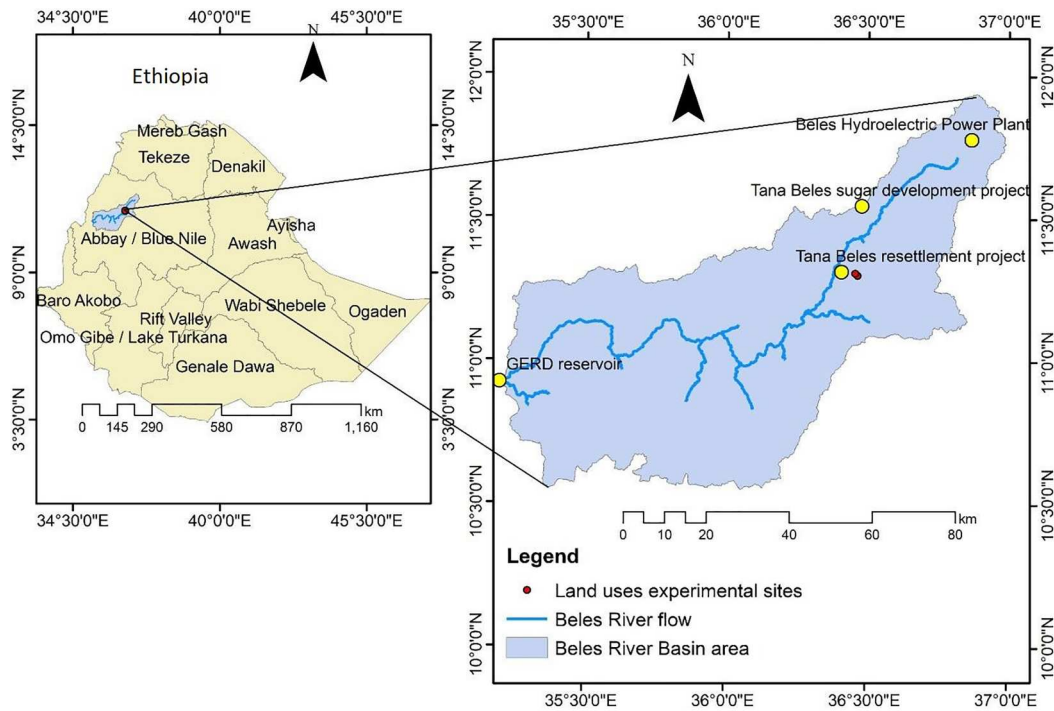


Figure 1. Location of the Beles resettlement and Villagization area (Gebre 2002).

Agro-ecology of the middle Beles River Basin is categorized into warm, moist lowlands and warm sub-humid lowlands (Abbute 1997).

The main farming system in the watershed is a crop-livestock mixed farming system and characterized by rain-fed agriculture. The dominant vegetation in the study area consists of dry combretum wooded, including *Acacia* species, Mango (*Mangifera indica*), Ethiopian olive tree (*Olea europaea subsp cuspidata*), water pear (*Syzygium guineense*), and neem (*Azadirachta indica*), Wanza (*Cordia africana*) (Nyssen *et al.* 2018). Moreover, traditional agroforestry includes lowland bamboo in boundary plantation, Mango with maize, and Papaya with maize. The dominant crops are grown in the study area including maize (Zeamays), Finger millet (*Eleusine coracana*), sorghum (*Sorghum bicolor L. Moench*), soyabean (*Glycine max*), rice (*Oryza sativa*) and groundnut (*Arachis*

hypogaea) of which soybean and groundnut are major cash incomes the remains for food consumption. Cattle, goats, poultry, and donkeys are major livestock in husbandry. In addition, on-farm activities such as daily laborer, petty trade, fishery and charcoal production also another alternative sources of income.

2.2. Experimental runoff plot setup and design

The experiment was conducted over two consecutive years (2021 and 2022) during the main rainy season. The study compared three land uses: CL, FG, and ZG. The runoff plots were arranged in a randomized complete block design with three replications on Nitisols. This design was used to minimize variation and ensure accurate estimation of experimental errors within the units (Hudson 1993), as illustrated in Table 1 and Figure 3(a–c). The average slope gradient

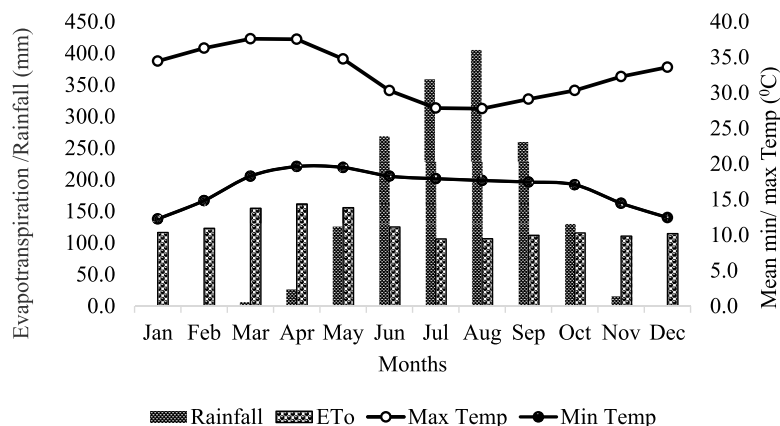


Figure 2. Middle Beles River Basin long-term climate data from the Pawe Agricultural Research Center for the period 1987–2022.

Table 1. Treatments in terms of land management practices for the study sites.

No	Treatments	Descriptions of treatments
1	Cultivated land	Farmers practiced conventional methods without supportive land management practices. The plot was intensively tilled using a local tool known as <i>Maresha</i> , which reaches a depth of 15–20 cm. In 2021, the plot was planted with finger millet, and in 2022, it was used for growing soybean.
2	Free grazing (Conventional grazing)	The land was historically used for conventional grazing, with runoff plots established without any supportive measures. Animals were allowed to graze freely, and the plots were not fenced, permitting unrestricted grazing throughout the study period.
3	Zero grazing (cut and carry system)	The land's history was similar to conventional grazing. We partitioned it and established runoff plots. After establishing the plots, livestock were not allowed to enter them during the research period. The grass was cut and carried at the end of November each year (Figure 10).

of the runoff plots was 9% with uniform soil type and aspect direction. A runoff plot was established at 22.6 m long (along the slope) and 4 m wide (along the contour line) (Wolka *et al.* 2021). All plots were bounded by hydrologically isolated corrugated iron sheets inserted into the ground 15 cm and above 35 cm to prevent lateral movement into and onto runoff, as shown in Figure 3. The experimental site is also free from the presence of rock fragment cover (Figure 3) because it affects soil erosion (Tilahun *et al.* 2024). A trapezoidal runoff collection trench was installed at the lower end of each runoff plot. The trench was lined a geomembrane plastic sheet with a thickness of 5 mm. The capacity of each collector trench was 5.4 m³, which is sufficient to accommodate runoff from extreme rainfall events. In the experimental area, a manual rain gauge was installed for the purpose of measuring the daily rainfall amount.

2.3. Runoff and soil loss measurement

At 8:00 am every 24 h, the runoff volume in each trench was measured. Prior to the collection of representative runoff samples, each runoff collector trench was thoroughly mixed. Subsequently, two-liter samples of runoff were collected using plastic bottles, one liter was used to determine the concentration of sediment (soil loss), while the remaining liter was employed to assess the sediment-associated outflow of SOC and nutrients. The total runoff accumulated in each trench was measured using a ten-liter plastic bucket and a one-liter graduated jug (Demissie *et al.* 2022, Kebede *et al.* 2023). Measurements were recorded for each rainfall event when runoff was observed and collected in the trenches. Once the runoff data were recorded, each trench was then

cleaned and emptied for the next day. Daily runoff samples were taken to the Pawe Agricultural Research Center for further analysis. During the experimental period, 693 runoff samples were collected to measure sediment concentration for the main rainy seasons of 2021 and 2022. Daily collected runoff samples were filtered through Whatman filter paper 42 Ashless cat No. 1442-125. First, the empty filter paper was weighed by a digital balance (0.001 g precision), and then the one-liter runoff sample was poured into the filter paper. The filter sediment was oven-dried at 105°C for 24 h. The oven-dried sediment was weighed, and then the weight of the filter was subtracted to get the actual soil loss. Finally, the sediment concentration was estimated from one liter of the runoff sample. The total soil loss was estimated by multiplying daily sediment concentration (g L⁻¹) 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. Runoff depth (mm) was calculated by dividing the runoff volume (liters) by the plot area (90.4 m²). Runoff coefficient and soil loss were calculated as described in equations (1) and (2) as described by (Belayneh *et al.* 2020).

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

$$\text{Total soil loss } t \text{ ha}^{-1} = \sum \frac{SC * V_r}{A_p} 10^{-6} \quad (2)$$

Where SC is sediment concentration (g L⁻¹), V_r is the total volume of runoff (in liters), and A_p is plot area (in square meters)

2.4. Soil data collection, analysis, and determination of SOC and nutrient losses

The soil data were collected from two sources: the runoff plot experiment test site and the sediment-associated nutrient losses. To evaluate the losses of sediment-associated SOC and nutrients from runoff plots, the study followed a similar procedure that was described above for soil loss determination. The only differences were that daily runoff samples were bulked into 30-liter jars (with three treatments $\times$ three replications), and a total of nine jars were prepared at the experimental sites. The bulk runoff samples were then decanted from the jars, and the remaining sediment was air-dried. Composite sediment from the seasons (June, July, August, and September) was labeled and taken to the soil laboratory for further analysis.

Additionally, soil samples were gathered from the experimental sites during the establishment of the runoff plots to obtain baseline data at depths of 0–20



Figure 3. Partial pictorial view of land uses and design, and data collection in the Middle Beles River Basin. (a) Cultivated land; (b) free grazing; (c) zero grazing; (d) cultivated post rainy season moisture collection; (e) zero grazing post rainy season moisture collection; (f) free grazing post rainy season moisture collection; (g) infiltration data collection, and (h), sediment filtration from runoff.

and 20–40 cm. A total of eight soil samples (two distributed and six undistributed) were initially collected from five sampling points per land use and mixed to create a composite sample. Following careful mixing of the point samples, approximately 1 kg of the composite sample was packed in a plastic bag and taken to the laboratory for further analysis. To determine bulk density, undisturbed soil samples were collected using a core sampler method with a height and inner diameter of 5 cm each (approximately volume = 98.125 cm³). The soil samples were air-dried in air soil preparation room, then ground using a ceramic mortar and pestle, and finally sieved through a 0.5 mm sieve (Jones 2001).

Soil particle size distribution was analyzed by the Bouyoucos hydrometer method (Bouyoucos 1962). Soil bulk density was calculated using 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).

Soil pH (pH meter) was determined 1:2.5 Soil: Water ratio potentiometric method (Van Reeuwijk 2002). The SOC content was determined (Walkley and Black) wet digestion method (Walkley and Black 1934). Total nitrogen was analyzed through the Kjeldahl digestion, distillation, and titration method (Bremner and Mulvaney 1982), while available P was

measured extraction solution of Bray II method (Bray and Kurtz 1945). The CEC was measured using the ammonium acetate method at pH 7.0 after extracting the soil samples (Houba *et al.* 1989).

The loss of SOC, N, and P associated with sediment was then calculated as described by in equations (3)–(5).

$$\text{SOC loss kg/ha} = \text{SOC sediment associated concentration in \%} \times 10 \times \text{soil loss kg/ha} \quad (3)$$

$$\text{N loss kg/ha} = \text{N sediment associated concentration in \%} \times 10 \times \text{soil loss kg/ha} \quad (4)$$

$$\text{P loss kg/ha} = \text{P sediment concentration in sediment mg/kg} \times 10^{-3} \times \text{soil loss kg/ha} \quad (5)$$

$$\text{Soil porosity (\%)} = \left(1 - \frac{\text{Bulk Density}}{\text{Particle Density}} \right) \times 100 \quad (6)$$

10 is a conversion factor to convert N and C into g kg^{-1} and 10^{-3} is the conversion of mg kg^{-1} of P into g kg^{-1} .

2.5. Measuring above and below-ground biomass of grasses and crops for the determination of carbon stock

Above-ground biomass of grasses and crops was collected during the post-rainy seasons. The study was assigned five quadrants per plot to representative sample areas (1 m × 1 m). All above-ground (standing and dropped) biomass was collected from the assigned quadrants in all land uses. Fresh biomass of grasses and crops from each quadrant was weighed using spring balances and recorded, as presented in Figure 10. Fresh sub-samples of biomass from each quadrant were measured using digital sensitive balances. Then, it was labeled and taken to the laboratory to determine the dry biomass. Belowground biomass of these grasses and crops were determined by multiplying the above-ground biomass weight by 0.27 as recommended by the IPCC (2006) using the root-to-shoot ratio of the tropical mountain system. In the laboratory, all sub-samples were oven-dried at 70°C for 48 h and weighed using a digital balance (precision = 0.001). The dried biomass was ground by an electronic grinding mill. Then, the powder was heated at 550°C for 6 h in a muffle furnace known as a loss on ignition (LOI) procedure as described by Nelson and Sommers (1996). Then, the dry mass of the sample per plot is calculated by dividing the sub-sample dry mass by the sub-sample fresh biomass and then multiplying the result by the total biomass of the plot sample.

Above-ground and below-ground biomass, along with soil organic carbon contents, were determined using equations (7)–(10) as recommended by (Pearson *et al.* 2005, Terefe *et al.* 2020).

$$\text{AGC} = \% \text{OCB} \times \text{MB} \quad (7)$$

$$\text{TBGC} = \% \text{OCB} \times \text{MB} \times \text{CF} \quad (8)$$

$$\text{TSOC} = \text{SOC\%} \times \text{BD} \times \text{D} \quad (9)$$

$$\text{TC} = \text{TAGC} + \text{TBGC} + \text{SOC} \quad (10)$$

Where TAGC = total aboveground carbon of biomass (t ha^{-1}), % OCB = total carbon content in leaves, stems, barks, and branches, as determined in the laboratory, MB = Oven-dried above-ground biomass, SOC = Soil organic carbon (%) determined in the laboratory, BD = Soil bulk density (g cm^{-3}), D = soil depth (cm), TC = total carbon stock for all carbon pools (t ha^{-1}), TBGC = Total below-ground carbon in biomass (t ha^{-1}), CF = conversion factor and SOC is soil organic carbon (t ha^{-1}).

To assess the CO₂ mitigation status under different land uses in the study area, the carbon stock was converted to tons of CO₂ equivalent. This conversion was performed using the equation provided by Terefe *et al.* (2020), as shown in equation (11).

$$3.6 \left(\text{molecular weight ratio of CO}_2 \text{ to O}_2 \frac{44}{12} \right) \quad (11)$$

Gravimetric techniques were employed to determine soil moisture content, as described by Hillel (1971), and a total of 180 soil moisture measurements were data collected during the experimental period. This method used an Edelman auger for sampling, and sub-samples were immediately weighed with 0.001 g precision at the site. The moist samples were then oven-dried at 105°C for 24 h, reweighed, and the mass of water lost was calculated as a percentage using the following equations.

$$\text{MC\%} = \frac{W2 - W3}{W3 - W1} \times 100 \quad (12)$$

Where W1 is weight of tin (g), W2 is weight of moist soil + tin (g), and W3 is weight of dried soil + tin (g)

2.6. Measurements of infiltration

Infiltration measurements were undertaken during the dry season using a double-ring infiltrometer. Three replicate measurements were taken in each land use type. The inner rings had a diameter of 30 cm, while the outer rings had a diameter of 60 cm. The rings were driven approximately 15 cm into the soil using a metal plate and a sledgehammer. The weather conditions during all measurements were sunny. The infiltration rate was monitored until it reached a stable value. After data collection, a functional relationship

between the water depth infiltrated and the elapsed time was established by calculating time intervals corresponding to successive water depth readings. The cumulative infiltration was then computed based on these measurements.

2.7. Statistical analysis

Before analyzing the data for normality, the Shapiro–Wilk, Bartlett, and Durbin–Watson tests were performed with AgRoR package (Shimizu *et al.* 2022). The units for runoff and soil loss were converted to mm and t·ha⁻¹, respectively. One-way analysis of variance (ANOVA) followed by least significant difference (LSD) tests were used to examine differences in infiltration capacity, carbon stock, sediment-associated nutrient losses, SOC, and nutrients across different land uses ($p < 0.05$). Two-way ANOVA with Tukey's honest significant difference (HSD) test was employed to assess differences in runoff, soil loss, and soil moisture content ($p < 0.05$). All statistical analyses were conducted using R software version 4.4.2.

3. Results

3.1. Soil characteristics of experimental land uses

The selected soil properties of the soil from the experimental runoff plots are presented in Table 2. The findings suggest that both land uses have clay-textured soils, with lower bulk density and higher porosity in CL compared to FG. This is likely due to soil intervention by tillage at CL, loosens the soil, decreasing bulk density, while livestock trampling in grazing areas compacts the soil over time, reducing pore space and increasing bulk density. Both land uses showed low levels of SOC and pH, along with moderate CEC, as classified by Hazelton and Murphy (2007) (Table 2). This is due to the high rainfall and temperatures in the study area, which promote rapid organic matter decomposition, depleting soil carbon stocks and releasing carbonic acid as a byproduct. This process can result in reduced soil carbon and added acidification. Low

SOC content implies reduced soil binding, resulting in a weaker soil structure that is more prone to removal by water. Therefore, understanding how soil properties interact is essential for predicting water absorption and the extent of surface runoff.

3.2. Effects of land uses on runoff and runoff coefficient

The total annual rainfall recorded at the experimental sites was 1402 mm in 2021 and 1254 mm in 2022. The proportion of annual rainfall that occurred between June and October was 88% in 2021 and 95% in 2022. During these months, rainfall was greater than evapotranspiration, potentially contributing to runoff generation in the study area. During the study period, there were 128 rainy days in 2021 and 139 in 2022. Although the total rainfall was higher in 2021, whereas the number of rainy days was greater in 2022, indicating notable variation between total rainfall and rainy days over the two years. The highest extreme daily rainfall was recorded on 29 June 2021 (55.2 mm) and 31 July 2022 (74 mm). The occurrence of extreme rainfall in June and early July induces the highest erosion rates as the soil surface remains exposed to direct rainfall until the crops are fully covered, resulting in high rates of soil removal.

Over the two rainy seasons, we evaluated annual runoff under different land uses (Figure 4). The total runoff ranged from 241 to 542 mm in 2021 and from 227 to 444 mm in 2022. In all land uses, the runoff values generated in 2021 were larger than in 2022, likely due to the significant variation in rainfall amounts between the two years. The maximum runoff value was observed under FG, while the minimum runoff value was observed in ZG. This result clearly shows the impacts of conventional livestock husbandry, where soil compaction and vegetation removal by livestock generated the highest runoff (Figure 3f). In plots treated with ZG, the runoff was the lowest (227–241 mm) due to erosive energy being intercepted and dissipated by vegetation and

Table 2. Soil characteristics before the establishment of the runoff plots at the experiment sites.

Soil parameters	Critical values and classification	Land use types				References
		Cultivated land		Grazing land		
		Soil depth in cm				
		0–20	20–40	0–20	20–40	
Clay		44	44	44	40	
Silt (%)		16	14	12	12	
Sandy (%)		40	42	44	48	
Textural class	Clay	Clay	Clay	Clay	Sandy clay	
BD (gcm ^{−3})	Low	1.21	1.23	1.33	1.34	Hazelton and Murphy (2007)
Soil porosity (%)	Moderate	54	50.9	49.8	49.3	Hillel (1971)
Soil pH	Low	5.43	5.44	5.32	5.36	Landon (1991)
SOC (%)	Very low	1.08	1.11	1.18	1.07	Landon (1991)
TN (%)	Low	0.14	0.16	0.14	0.14	Hazelton and Murphy (2007)
P (mg kg ^{−1})	Low	1.08	1.11	3.07	2.05	Hazelton and Murphy (2007)
CEC (cmol (+) kg ^{−1})	Medium	18.94	17.93	18.07	22.01	Landon (1991)

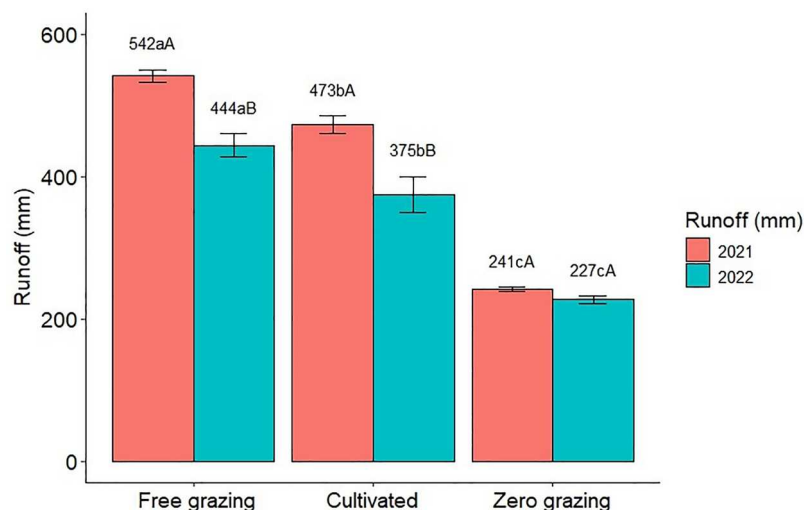


Figure 4. Runoff effects under different land uses in the middle Beles River Basin over two years (2021–2022). Means followed by the same lowercase letter between treatments and the same uppercase letter over the year do not differ significantly as determined by the Tukey test ($p < 0.05$).

grass cover (jangle grasses (≥ 2 m height)), giving a low chance to produce runoff (Figure 3e). ANOVA results (Figure 4) showed that land uses had a significant effect on runoff generation ($p < 0.05$), in the order of $ZG < CL < FG$ over the two years. This disparity is largely attributed to multiple factors, particularly within the free-grazing system, where extensive grazing depletes grass cover and livestock trampling compacts the land surface. As a result, the soil became bare and highly compacted. Subsequently, rainfall hitting the compacted soil surface generated high runoff. On the other hand, low-rate water enters the soil. Similarly, CL faces high soil erosion due to vegetation removal and frequent tillage, which weakens soil structure and reduces organic matter. Moreover, insufficient cover crops (before crop establishment) further increase runoff and soil detachment. This variation was attributed to the cons and pros of human-induced activities on land uses and variability in rainfall amount and distribution. ZG showed the highest runoff control efficiencies due to minimal topsoil disturbance, attributed to the grass barrier's and surface soil cover, which enhances infiltration rates.

Estimating runoff coefficients (RC) is crucial for understanding how much rainfall is converted into surface runoff and determining which land uses contribute the most. This knowledge is essential for designing runoff control measures, estimating water yield, and forecasting flood risks (Marzluff *et al.* 2008). Initially, RC values were similar across all land uses due to low antecedent moisture, but distinct trends emerged during the rainy season (Figure 5). Lower slopes demonstrated greater runoff prevention effectiveness, whereas higher slopes retained significantly less water (Figure 5). FG showed the highest mean annual RC at 37%, whereas ZG, characterized by dense savanna grass cover, showed the lowest

mean annual RC at 17.6%, indicating more effective runoff management. Human-induced activities in both CL and FG were found to significantly influence RC patterns. Therefore, site-specific and land use and management specific RC evidence is crucial to implement appropriate farm water management practices and for generating model input parameters for water resource management and making informed decisions (Assaye *et al.* 2024, 2025).

3.4. Effects of land uses on soil loss

Soil loss results in 2021 and 2022 are presented in Figure 6. In 2021, soil loss was recorded at 48.5 t ha^{-1} for CL, 12.5 t ha^{-1} for FG, and 2.6 t ha^{-1} for ZG. In 2022, these values had decreased to 41.8 t ha^{-1} for CL, 9.7 t ha^{-1} for FG, and 2.1 t ha^{-1} for ZG (Figure 6), corresponding to a reduction in total rainfall in the second year. The highest soil loss was observed in CL, while the lowest was recorded in ZG, followed by FG (Figure 6). These results indicate that human-induced activities, such as frequent tillage practices for seedbed preparation, played a greater role in soil loss compared to the impact of vegetation removal or soil disturbances from livestock encroachment. ANOVA results (Figure 6) revealed significant differences ($p < 0.05$), with decreasing soil loss in the order of $CL > FG > ZG$ throughout the study period. Soil loss under CL showed significant variation over the years ($p < 0.05$); whereas no significant differences were observed in other land uses ($p > 0.05$). The notable variability in CL is attributed to the presence of different cover crops during the study period. In 2021, CL was covered with finger millet, while in 2022 it was planted with soybean, and these crops likely had different leaf area indices, which could have affected soil loss.

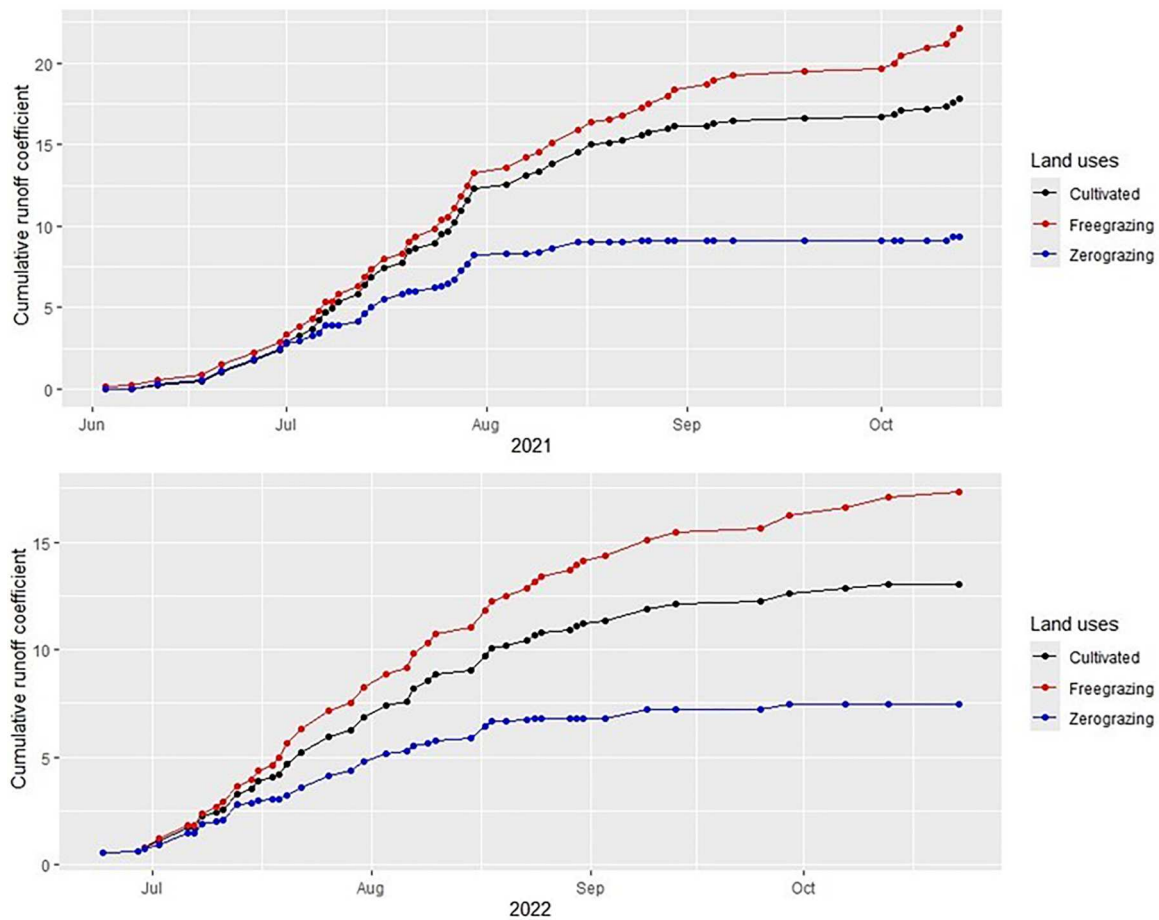


Figure 5. The relations between cumulative runoff coefficients and daily rainfall across different land uses for two rainy seasons.

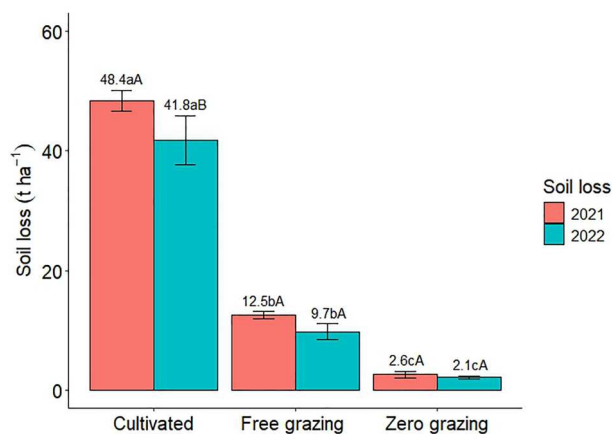


Figure 6. Soil loss results under land uses in the middle over two years (2021–2022). Means followed by the same lower-case letter between treatments and the same uppercase letter over the year do not differ significantly as determined by the Tukey test ($p < 0.05$).

3.5. Sediment-associated nutrient and soil organic carbon losses in response to land uses

Soil erosion not only results in the loss of soil constituents but also the outflow of essential soil nutrients and organic carbon, primarily through water erosion. The loss of sediment-associated nutrients (N and P) and SOC varied significantly across

different land uses (Table 3). The mean annual losses were 1139.6 kg ha⁻¹ for SOC, 78.3 kg ha⁻¹ for N, and 0.9 kg ha⁻¹ for P. The highest losses in CL were recorded at 1268 and 1011 kg ha⁻¹ for SOC, 86.4 and 70 kg ha⁻¹ for N, and 0.9 and 0.8 kg ha⁻¹ for P in 2021 and 2022, respectively. In contrast, ZG recorded the lowest mean annual losses at 70.2 kg ha⁻¹ for SOC, 4.6 kg ha⁻¹ for N, and 0.7 kg ha⁻¹ for P. SOC and N losses in CL were more than sixteen times higher than losses in ZG, and three times higher than losses in FG. These differences in SOC and nutrient losses were highly significant ($p < 0.05$, Table 3), in the order of CL > FG > ZG. The results indicate that SOC and nutrient losses were more significantly affected by human-induced factors than by livestock grazing impacts. CL experienced maximum SOC loss through erosion, aggravated by high precipitation and optimal temperatures, compounded by intensive human activities that accelerate organic matter decomposition. This process is made worse by mechanical tillage, which speeds up aeration and oxidation rates. This makes organic matter more prone to erosion. This significant SOC removal may contribute to greenhouse gas emissions and disrupt ecosystem services. Similarly, N was also removed in large quantities from CL, as crops require these nutrients in substantial amounts, making them limiting

Table 3. Sediment-associated losses of nutrients (N and P) and SOC from FG, ZG and CL in the middle Beles River Basin.

Land uses	Sediment-associated loss of SOC, N, and P								
	SOC (kg ha ⁻¹)			N (kg ha ⁻¹)			P (kg ha ⁻¹)		
	2021	2022	Mean	2021	2022	Mean	2021	2022	Mean
CL	1268.3 ^a	1010.8 ^a	1139.6 ^a	86.3 ^a	70.2 ^a	78.3 ^a	0.9 ^a	0.8 ^a	0.9 ^a
FG	482.2 ^b	286.4 ^b	384.3 ^b	30.1 ^b	17.4 ^b	23.7 ^b	0.8 ^a	0.7 ^b	0.8 ^b
ZG	82.9 ^c	57.6 ^b	70.2 ^c	5.7 ^c	3.5 ^b	4.6 ^c	0.7 ^b	0.7 ^b	0.7 ^c
CV %	7	22.9	24.2	5.5	20.6	22.8	4.2	1.8	6
0.05	**	**	**	**	**	**	**	**	**
LSD (5%)	97	234.8	160.6	5.1	14.2	10.1	0.08	0.03	0.06

CL = Cultivated land, ZG = Free grazing land, ZG = Zero grazing land and CV = Coefficient of variation and LSD = least significant difference. Means in the same letter are not significantly different at 5% level of probability; ** = highly significant different at $p \leq 0.01$.

factors compared to other nutrients. As a result, agricultural productivity may be compromised unless measures are taken to prevent this deterioration. The study also reported considerable variation in available P among land uses, with generally low available P in the study area (Table 2).

3.5. Temporal variability of soil moisture content

The impact of land use, seasons, and soil depth on soil moisture content (SMC) was monitored over a two-year period (Figures 7 and 8). Although the variability of SMC had been observed in relation to land use, ZG generally had the highest value at 24.1 mm, followed by CL at 22.6 mm and FG at 22.4 mm, respectively. These variations in SMC are likely due to factors related to land surface interference by human or livestock activities and areas excluded from both. Over the study period and across seasons, SMC ranged from 7% to 44% in CL, 8% to 42% in FG, and 9.0% to 44% in ZG. In all land uses, the highest mean SMC values were recorded on 17 July 2022 (43%), followed by 12 October 2022 (34%), whereas the lowest values were

recorded on 23 January 2023 (14%), and 4 April 2023 (8%). These results clearly show seasonal variability of SMC over season, with excess SMC during summer associated with high rainfall events (Figure 2) and deficiency in winter and spring due to high evaporation (Figure 2). Optimal moisture levels were observed in the early rainy season (May) and the post-rainy season (October to November). Since the study area is rainfed agriculture, these conditions are ideal for seedbed preparation and planting, and crop growth.

As shown in Figures 7 and 8, SMC was significantly ($P < 0.05$) influenced by the combination of land use, season, and soil depth. Figure 8 further illustrates that SMC also showed significant variations ($P < 0.05$) across seasons and soil depths. SMC generally increased with soil depth. However, the influence of seasons on SMC varied considerably by soil depth, with strong interaction effects observed in November, December, and January.

3.6. Infiltration capacity under land use

A cumulative comparison curve was presented in Figure 9, with cumulative infiltration/water intake

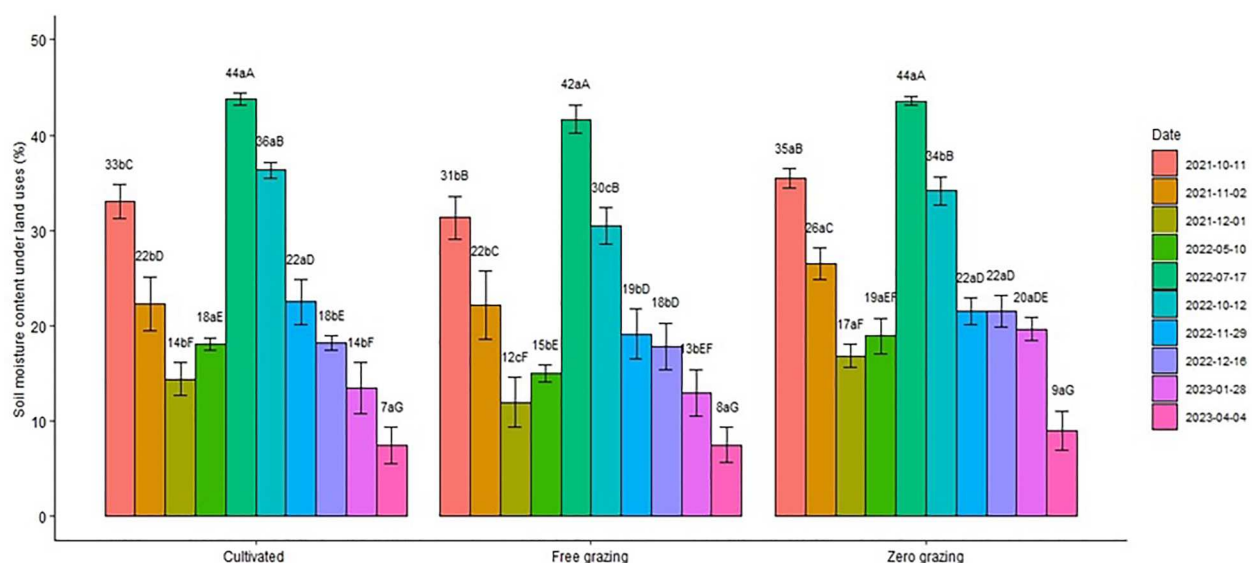


Figure 7. Variability of soil moisture under different land use over the season in the study area Means followed by the same lowercase letter between treatments and the same uppercase letter over the year do not differ significantly as determined by the Tukey test ($p < 0.05$).

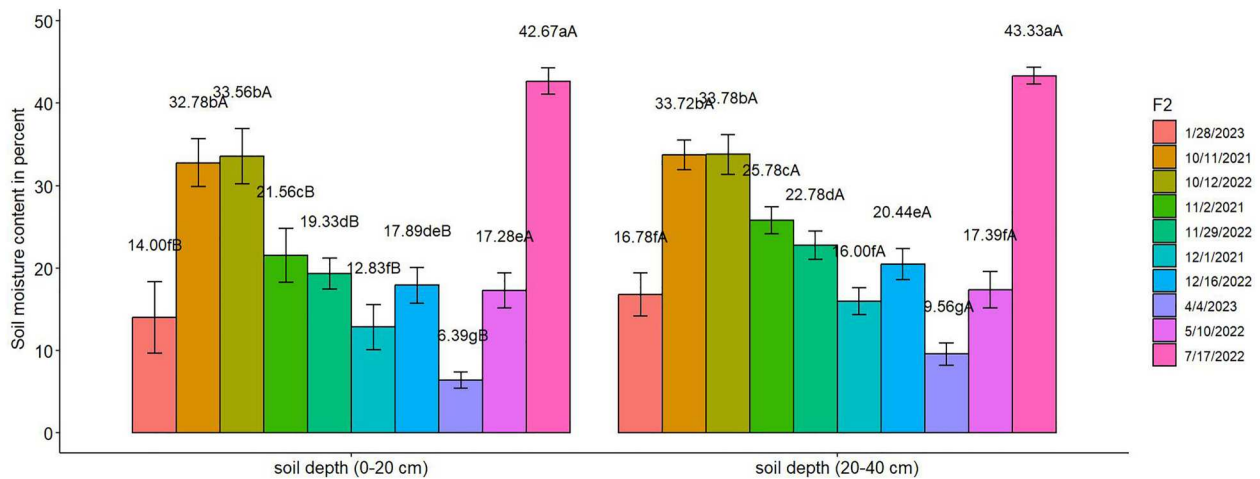


Figure 8. This figure presents soil moisture variability across two depth intervals (0–20 and 20–40 cm) in the study area. Means followed by the same lowercase letter between treatments and the same uppercase letter over the year do not differ significantly as determined by the Tukey test ($p < 0.05$).

(cm) on the y-axis and cumulative time taken (hour) on the x-axis. The results indicate that infiltration capacity of CL was much more than the FG and zero ZG. This implies that CL demonstrates faster and high infiltration capacity. Whereas, ZG uses showed lowest infiltration but longer infiltration period. Free grazing land uses have measured low infiltration capacity and not remain for long period most likely due to compaction induced by livestock trampling. The infiltration capacity of cultivated

lands was significantly higher (150 cm) followed by, FG (33.8 cm) and ZG land us (14.7 cm) as presented [Figure 9](#). This is because the CL was free from plough pan formation, as it had not been plowed for an extended period and had not experienced livestock compaction. In addition, the CL is characterized by the prevalence of termite mounds, which are constructed beneath subsoil that are networked. This may contribute to a superior infiltration capacity compared to other land uses.

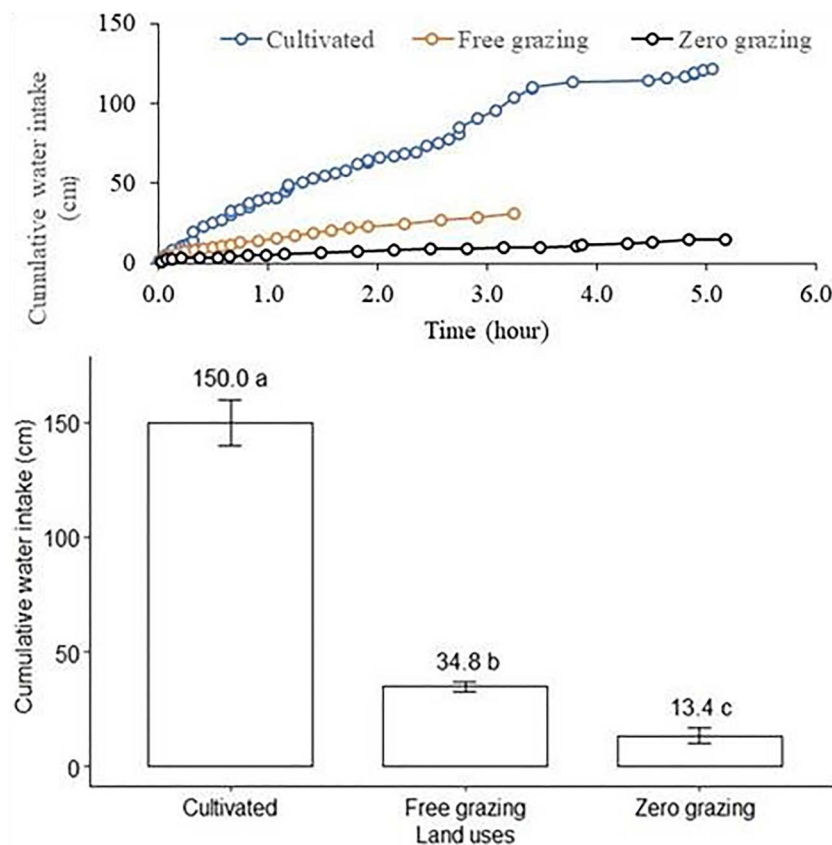


Figure 9. Cumulative infiltration curves for land use types (data collected on 23 January 2022).

The ZG showed lower water infiltration capacity but required more time to reach a steady state compared to FG. The soil surface was covered by grass, which reduced evaporation and improved soil moisture content in this treatment. For example, on 23 January 2022, we observed the highest soil moisture content in the zero-grazing plots (Figures 7 and 8). In contrast, FG had lower water retention due to compactness and experience higher surface runoff. Moreover, it was more exposed to sunlight, resulting in higher evaporation. Consequently, the FG had the lowest SMC among land uses (Figure 8), thereby allowing it to absorb more water. Under different land uses infiltration rate and cumulative infiltration significantly varied ($P < 0.05$) values from under CL had $> FG > ZG$.

3.7. Carbon stock under different land uses

The estimation of carbon stocks, including above- and below-ground biomass and SOC under different land uses, was estimated and is presented in Table 4 and Figure 9(a–c). Above-ground carbon stocks were highest in ZG, followed by CL and FG, with values of 9.8, 4.7, and 1.9 t ha⁻¹, respectively. Similar mode was observed below the ground carbon stocks, with the highest in ZG (2.6 kg C ha⁻¹) and the lowest in FG (0.5 kg C ha⁻¹). ZG had five times the carbon stock of FG and double that of CL. Consequently, ZG was the most remarkable contributor to climate change mitigation among the land uses. SOC showed significant variations ($p < 0.05$) under land uses and soil depths (Figure 9 and Table 4). SOC was highest under FG, with 51.8 t C ha⁻¹ at a depth of 0–20 cm and 44.2 t C ha⁻¹ at a depth of 20–40 cm. In comparison, under CL, SOC was lower, with 44.2 t C ha⁻¹ at a depth of 0–20 cm and 33.7 t C ha⁻¹ at a depth of 20–40 cm. The lower carbon stock variation observed in CL was attributed to continuous tillage practices, which accelerate decomposition rates (via

oxidation) and expose the soil to human-induced water erosion. In contrast, FG performed better than CL due to its extensive root systems and undisturbed soil, which enhance its capacity to capture and store atmospheric carbon dioxide.

Carbon dioxide equivalent (CO₂eq) values followed similar modes, with ZG recording the highest at 386.8 t CO₂eq ha⁻¹, followed by FG at 340.3 t CO₂eq ha⁻¹, and CL at 307.7 t CO₂eq ha⁻¹. Statistically, these differences were highly significant ($p < 0.001$) across land uses, highlighting the influence of land-use practices on carbon sequestration potential.

4. Discussions

4.1. Runoff and soil loss responses under different land uses

The highest mean annual runoff was observed in FG, followed by CL and ZG, with values of 493, 424, and 234 mm, respectively. Likewise, the greatest soil loss was recorded in CL, followed by FG and ZG, with values of 45.1, 11.1, and 2.4 t ha⁻¹, respectively (Figures 4 and 6). In line with these findings, runoff and soil loss were recorded in FG (280.7 mm, 32.9 t ha⁻¹) and CL (258.7 mm, 45.6 t ha⁻¹) in the highlands of Ethiopia (Adimassu *et al.* 2020). Yaekob *et al.* (2022) found that runoff was 24% greater in FG than in CL, and soil loss was 57% higher in CL compared to GL. Similarly, Ebabu *et al.* (2023) found runoff and soil loss values of 593 mm and 1.5 t ha⁻¹ in FG and 450 mm and 71 t ha⁻¹ in CL in the UBNB. In contrast to our findings, some studies suggest that soil loss and runoff were higher from FG (1280 mm, 26.5 t ha⁻¹) than from CL (108 mm, 7.0 t ha⁻¹) at Mayleba (Ebabu *et al.* 2023). Taye *et al.* (2013) also reported that in northern Ethiopia, soil loss and runoff were 83% and 71% higher from FG compared to CL. These variations in runoff and soil loss are likely attributable to a number of factors, including the degree of human activity, climate variability, soil lithology, and topography (Desta *et al.* 2021). Elsewhere in China, Cheng *et al.* (2016) reported that open grazing was a major driver of soil erosion, leading to reduced soil productivity. In Europe and the Mediterranean, Maetens *et al.* (2012) reviewed studies that found higher mean soil loss rates from CL compared to rangeland.

Our findings indicate that the soil loss under CL (45 t ha⁻¹) was exceeds the acceptable limit of the mean soil loss 10 t ha⁻¹ as suggested by Hurni (1983) for Ethiopian condition. Our finding demonstrates an urgent need to address the problem of soil erosion on CL and required various soil erosion controlling techniques such as mechanical, vegetative and agro-nomic land management measures to reduce soil loss to an acceptable level.

Table 4. Mean values of above and below ground carbon and soil carbon under different land uses in the study.

Land use type	Ecosystem Carbon stock (t C ha ⁻¹)					
	Above ground carbon	Below ground carbon	Soil depths in cm			CO ₂ equiv.
			0–20	20–40	Total	
Zero grazing	9.8 ^a	2.6 ^a	51.8 ^a	42.2 ^a	104.4 ^a	386.8 ^a
Free grazing	1.9 ^c	0.5 ^c	50.8 ^a	39.3 ^a	92.7 ^b	340.3 ^b
Cultivated	4.7 ^b	1.3 ^b	44.2 ^b	33.7 ^b	83.7 ^c	307.7 ^c
CV	29.5	28.96	2.5	10.1	4.3	4.3
0.05	**	**	**	**	**	**
LSD	2	0.5	1.5	4.7	5	18.5

CV = coefficient of variation and LSD = least significant difference. Means the same letter are not significantly different at 5% level of probability * = significantly different at $p \leq 0.05$; ** = highly significant different at $p \leq 0.01$; and *** = highly significant different at $p \leq 0.001$.

As displayed in [Figures 4 and 5](#), the levels of runoff and soil loss were reduced by 53% and 78%, respectively, in ZG in comparison to FG. Supporting our findings, runoff and soil loss were lower in areas of restricted grazing (455 mm, 0.6 t ha⁻¹) than in FG areas (593 mm, 1.5 t ha⁻¹) in the humid highlands of Ethiopia ([Adimassu et al. 2014](#), [Ebabu et al. 2023](#)). [Mwendera and Saleem \(1997\)](#) also noted that controlled grazing led to a significant decrease in runoff and soil erosion when compared to unrestricted grazing practices. The results clearly demonstrate that excluding livestock preserved water and soil resources on-site while enhancing productivity. These findings offer valuable insights for policymakers and land managers engaged in planning, implementing, and evaluating sustainable land use, soil and water conservation strategies in warm, sub-humid lowlands and other similar agroecological settings.

4.2. Nutrient and SOC losses under different land uses

The analysis indicates that land use exerted significant influence on the loss of nutrients and SOC through soil erosion. Specifically, CL showed notably higher SOC (1139.6 kg ha⁻¹) and nutrient losses of N (78.3 kg ha⁻¹) and P (0.9 kg ha⁻¹) compared to other land use types ([Table 3](#)). Recent studies by [Adimassu et al. \(2020\)](#) and [Girmay et al. \(2009\)](#) highlighted that sediment-associated losses of SOC, N, and P were more marked in CL and FG land uses, while the lowest in ZG, due to strategic land management practices such as enclosure from animal interference and the presence of protective grass cover ([Figures 3e](#)). These measures effectively intercepted raindrops and dissipated erosive energy, resulting in reduced erosion. These results in line with previous studies by [Ebabu et al. \(2019, 2023\)](#) and [Sultan et al. \(2018\)](#), which also reported minimal nutrient losses. Likewise, [Cheng et al. \(2016\)](#) observed that grazing exclusion significantly enhanced SOC, N, and P levels in China. In India, however, higher losses of SOC, N, and P were noted in areas with open grazing and rainfed agriculture, as reported by [Kothyari et al. \(2004\)](#). Without effective interventions, soil erosion contributes to approximately 6.3%, 7% and 22% of annual nutrient losses of SOC, N, and P, respectively, with CL vulnerable due to inadequate land management practices. Thus, uncontrolled soil erosion can aggravate nutrient deficiencies, underlining the critical importance of sustainable land management strategies.

Soil organic carbon plays a vital role in enhancing soil health, carbon sequestration, and promoting soil biodiversity ([Lehmann and Kleber 2015](#), [Lal 2016](#)). Conventional tillage practices, by breaking soil aggregates and removing vegetation, lead to accelerated on-site SOC loss ([Lal 2004](#), [Van Oost et al. 2007](#)). This

depletion and redistribution of SOC can ultimately result in greenhouse gas emissions and disrupt the balance of ecosystem services. Poor agricultural practices in Ethiopia result in an annual loss of 3.7 t ha⁻¹ ([Van Beek et al. 2019](#)). Globally, soil erosion strips away nearly 1 billion tons of SOC each year, contributing 10%–15% to greenhouse gas emissions ([Lal 2018](#)). Therefore, mismanaged agricultural land use has the potential to worsen climate change if not appropriately addressed.

In our study, the mean annual N loss rate was 78.3 kg ha⁻¹ in CL. Supporting our findings, [Mulualet al. \(2024\)](#) reported that 47 kg ha⁻¹ of N loss was attributed to soil erosion, significantly influenced by the agroecosystem of UBNB. Similarly, [Wolka et al. \(2021\)](#) observed that between 38 and 77 kg ha⁻¹ of N was lost through soil erosion from areas under conventional tillage in southern Ethiopia. In central Ethiopia, N losses from CL and FG were reported as 48 and 33 kg ha⁻¹, respectively ([Adimassu et al. 2017, 2020](#)). In Kenya, [Okeyo et al. \(2014\)](#) documented an N loss rate of 18 kg N ha⁻¹. Our findings, along with those of other studies, clearly highlight that soil erosion, primarily driven by water, is a significant factor contributing to N loss. Our results clearly demonstrate that soil erosion through water the prominent cause of N loss and erosion controlling techniques may a key strategy for prevent losses this nutrient.

Our finding on P loss (0.9 kg ha⁻¹) from CL is in line with those reported in other studies. [Wolka et al. \(2021\)](#) documented P losses ranging from 0.1 to 1 kg ha⁻¹ in Southern Ethiopia, while in the Northwest lowlands of Ethiopia (Dibate), [Mulualet al. \(2024\)](#) reported losses of 0.5 kg P ha⁻¹ and [Girmay et al. \(2009\)](#) found 0.2 kg P in Northern Ethiopia. Studies from various countries have shown P losses due to soil erosion, such as 12 kg P ha⁻¹ in Kenya ([Okeyo et al. 2014](#)), 0.5 kg P ha⁻¹ in Mediterranean agroecosystems ([Martínez-Mena et al. 2020](#)), and 0.25 kg P ha⁻¹ in Europe ([Muntwyler et al. 2024](#)). Generally, P losses are lower compared to SOC and N losses, possibly due to P is immobile nature and its tendency to bind with aluminum, iron, or calcium oxides ([Johan et al. 2021](#)). According to [Mulualet al. \(2024\)](#), water erosion accounts for 46%–67% of nutrient losses at UBNB. Therefore, implementing water erosion mitigation measures is crucial to minimize the exportation of nutrients and SOC.

4.3. Soil moisture content, infiltration rate and carbon stock

The soil moisture content (SMC) showed significant variation ($P < 0.05$) across different land use types, seasons, and soil depths, with notable interactions among these factors ([Figures 8 and 9](#)). ZG showed the highest

SMC levels, whereas the SMC in CL and FG varied depending on the season and surface cover. This finding is consistent with Yimer *et al.* (2015), who reported higher SMC in exclosures (4.03%) compared to open grazing areas (3.7%) in the Arsi area of Ethiopia. Similarly, Selassie and Ayanna (2013) noted significantly higher SMC in grasslands (35.3%) compared to CL (31.8%) in northwest Ethiopia. Collectively, these results suggest that in CL, frequent tillage practices gradually deteriorate soil structure and organic matter, while the removal of crop residues further diminishes in soil water-holding capacity.

The lower SMC in CL may adversely affect rainfed agricultural crop growth and lead to reduced crop yields. Notably, SMC tends to increase with soil depth across different land uses and seasons, with statistical significance ($p < 0.05$). Fu *et al.* (2003) also observed that subsurface soil generally retains higher moisture content compared to the lower layers. In China, Niu *et al.* (2015) recorded lower soil moisture content under various land uses during the driest season. Understanding the response of SMC to different land uses is crucial for modeling soil hydrological processes and managing agricultural water effectively within the watershed.

In Figure 8, the infiltration capacity was notably higher ($p < 0.05$) in CL followed by FG and ZG. This discrepancy may be due to the data collection during the dry season (on 23 January 2022), where the land surfaces were devoid of organic materials and exposed to sunlight in both CL and FG, resulting in more efficient water intake compared to ZG. Consistent with our findings, Yimer *et al.* (2008) also documented higher infiltration capacity in cultivated land compared to open grazing areas. However, Mwendera and Saleem (1997) reported higher steady-state infiltration rates in ZG than in FG. This variation could be attributed to seasonal differences; our data were gathered at the end of January (dry season), whereas the previous authors conducted their research in October.

According to Table 4 and Figure 10(a–c), the highest carbon stock was observed in ZG, followed by CL, with values of 12.4, 6.4, and 2.4 kg ha⁻¹, respectively, estimated from both above- and below-ground biomass. This outcome is in line with findings by Terefe *et al.* (2020) who reported higher organic carbon stocks in treated grazing compared to open grazing and cropland in Central Ethiopia. Similarly, research in Eastern Ethiopia showed a notable decline in soil organic carbon in open grazing areas (1.59%) compared to exclosures (2.23%), attributed to intensive grazing (Abdulahi *et al.* 2021). Soil organic carbon levels at specific soil depths consistently favor ZG over FG and CL, at depths of 0–20 cm and 20–40 cm. Previous studies in the Tigray Region by Mekuria (2013) revealed altered soil properties in exclosures, with the highest soil carbon stocks under ZG. However, findings from arid areas in the Sekota areas of northern Ethiopia, as reported by Eshetie *et al.* (2024), contradicted our results. Our study indicates a decrease in carbon stock as soil depth increases, as detailed in Table 4, while their research suggests an increase. This disparity might stem from variations in moisture regimes, soil characteristics, soil biota, and root structures. Research conducted in China highlights that excluding free grazing can effectively enhance carbon stocks above and below ground and boost soil carbon sequestration rates (Deng *et al.* 2017). Therefore, ZG practices can foster biomass accumulation, carbon capture from biomass, and carbon sequestration in the soil.

5. Conclusion and recommendations

The current status of soil erosion, surface runoff, sediment-associated nutrient and SOC, and carbon stock middle Beles River Basin was evaluated under various land uses. The maximum losses of soil, SOC, N, and P were observed under CL, with mean values of 45 t ha⁻¹, 1139.5 kg ha⁻¹, 78.3 kg ha⁻¹, and 0.9 kg ha⁻¹, respectively. These losses were primarily due to accelerated soil erosion by water. As a result, the soil loss



Figure 10. The biomass status during harvesting time; Free grazing (a), zero grazing (b), and cultivated land (c).

rate was considerably higher than accepted mean soil loss tolerance of 10 t ha^{-1} for Ethiopian conditions. These findings highlight the pressing need for remediation through the implementation of sustainable land management practices that can either maintain or enhance ecosystem services in agricultural lands within the watershed.

The highest runoff and cumulative runoff coefficients were recorded for the FG land use, followed by the CL land use. This excess surface water outflow has significant synergistic effects, resulting in upstream topsoil loss, reduced soil fertility, lower crop yields, land degradation, and decreased groundwater recharge. Downstream, these effects manifest as flash flooding, increased sedimentation, damage to irrigation infrastructure, altered stream flow direction, and compromised stream bank stability.

The mean soil loss value under ZG was 2.3 t ha^{-1} , which was much below the soil loss tolerance threshold of 10 t ha^{-1} suggested by Hurni (1983) for Ethiopian conditions. Therefore, based on our findings, alternative grazing land management practices such as ZG (cut and carry), rotational grazing, and strip grazing could sustainably manage grazing land ecosystem services without relying on mechanical erosion control techniques in warm sub-humid lowland agroecology.

The findings of this study demonstrate that the influence of land use on runoff and soil loss, including SOC and nutrients, exhibits considerable variability. It is of paramount importance to conduct site-specific experimentation in order to develop accurate input parameters for models such as the Soil and Water Assessment Tool (SWAT). This is crucial for the effective management of agricultural resources, the planning of water resources, and the protection of environmental, as it enables the estimation of impacts and the formulation of informed strategies for the mitigation of these effects.

The study provides valuable insights for policymakers and land management practitioners seeking effective strategies to address soil erosion, nutrient loss, climate change, and water management in rainfed areas. The implementing sustainable land management practices for agricultural land use is of a paramount importance for the maintenance of and enhancement of ecosystem services within the watershed.

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Author contributions

Yenesew Assaye: Conceptualization, Investigation, Data collection, Methodology, Software, Data analysis, and Writing-original draft. **Gizaw Desta:** Conceptualization, Investigation, Methodology, Supervision, Data analysis, Writing Review, and Editing. **Eyayu Molla:** Conceptualization, Methodology, Supervision, Writing, Review, and Editing. **Zenebe Adimassu:** Methodology, Supervision, Writing, Review, and Editing; and **Mengistie Kindu:** Methodology, Data analysis, review, and editing.

Disclosure statement

No potential conflict of interest was reported by the author(s).

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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