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Monitoring soil quality of different land use systems: a case study in Suha watershed, northwestern highlands of Ethiopia

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Abstract

The problem of soil quality degradation has been becoming more severe in the highlands of Ethiopia due to soil erosion; land use and land cover change, and poor land management. The level of soil quality degradation was not well known and documented in the study area and the results of this study could provide new information to improve soil conditions. The present study was conducted to evaluate soil quality in terms of its physical and chemical fertility under different land use types in the Suha watershed, northwestern highlands of Ethiopia. A total of 27 composite surface soil samples (0–30 cm) were collected from adjacently located land-uses in three replications from two elevation gradients. Standard procedures were followed to analyze selected soil physical and chemical quality indicators. The differences in the mean values of the parameters were tested using a two-way analysis of variance. In addition, Soil Quality Degradation Index was evaluated to see the direction and magnitude of change in soil quality indicators. The analysis of variance results revealed that soil quality indicators such as index of soil aggregate stability, organic carbon (OC), total nitrogen (TN), and C:N ratio were significantly decreased in the cultivated land use system compared to other land use systems. On the other hand, the content of available Phosphorus was significantly higher in the cultivated land. Soil quality deterioration index values were highly negative for SOC (–71.3%) and TN (–67.7%) in the cultivated land, followed by grazing land (SOM = –35.5% and TN = –27.7%). Aggregated Soil Quality Index values also indicated that the status of soil quality under cultivated fields is rated as low, grazing land as optimal, and forest land as high. Generally, results indicated that land use and cover changes had adverse effects on soil quality indicators. Hence, soil management strategies, mainly Integrated Soil Fertility Management which integrates soil and water conservation strategies, are required to alleviate the problem of soil quality deterioration and improve agricultural productivity.

Keywords Soil quality, Land use type, Soil Quality Index, Suha watershed

Introduction

Soil degradation, manifested by the deterioration of soil's physical, chemical, and biological properties, is a global environmental problem that affects agricultural productivity, food security, climate change, and biodiversity (Young et al. 2015; Dagnachew et al. 2019). According to Nkonya et al. (2016), the negative consequences of soil degradation have resulted in a 60% reduction in the ecosystem services provided by soil, which now cost the globe between 6.3 and 10.6 trillion dollars annually. In Sub Sahara Africa (SSA), soil degradation has threatened

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the livelihood of the population since 67% of the land resource is degraded in different severity classes (Sileshi et al. 2019). In these countries, the major soil constraints include nutrient depletion, soil erosion, Al toxicity, low soil depth, high P-fixation, low CEC, salinity, and vertic properties (Zingore et al. 2015). Soil degradation is more severe in Ethiopia, and nutrient mining is the highest compared to SSA (Teferi et al. 2016). Soil degradation in the form of soil erosion and loss of soil fertility in the highlands of Ethiopia causes a reduction in agricultural productivity (Gashaw et al. 2014; Meseret 2016), which in turn impacts the livelihoods of the rural community and the economy of the country (Kassa et al. 2013; Adugna and Abegaz 2015). The study of Zingore et al. (2015) revealed that negative nutrient balances of macronutrients (N=40–70 kg/ha, P=7–10 kg/ha, and K=33–50 kg/ha) were the highest in Ethiopia compared to sub-Saharan Africa (SSA).

Soil quality can be defined as the capacity of soil to provide ecosystem services in a natural or managed environment (Andrews et al. 2004). This parameter cannot be measured directly; rather, it is determined based on soil quality indicators (physical, chemical, and biological properties) (Zornoza et al. 2015). Land use and land cover changes are the major drivers of soil quality deterioration and other environmental changes like soil erosion, climate change, loss of biodiversity, the carbon cycle, and impairment of ecosystem services (Wang et al. 2016; Singh and Benbi 2018; Lepcha and Devi 2020). In Africa, soil quality deterioration has been aggravated due to population pressure and subsistence farming associated with poor land use systems, which in turn has resulted in declining agricultural productivity and food insecurity (Obalum et al. 2012; Alarima et al. 2020). Previous studies conducted in Ethiopia and elsewhere indicated that land use and land cover changes have detrimental effects on soil quality indicators (Tsefahunegn 2016; Delelegn et al. 2017; Pham et al. 2018). Changes from natural forest to agricultural land are the major causes of extensive soil erosion, soil quality deterioration, depletion of soil nutrients, and declining agricultural production in the highlands of Ethiopia (Aredehey et al. 2019; Molla et al. 2022). Dagnachew et al. (2019) and Teferi et al. (2016) also reported that key soil quality indicators are significantly affected due to human-induced activities in the Upper Blue Nile basin. The highlands of Ethiopia faced critical soil-related problems from the perspective of soil quality and fertility. These include extensive soil erosion, soil acidity that covers about 40% of cultivated fields, a high rate of soil organic matter loss, and depletion of macronutrients (high rate of negative nutrient balances). Addressing the problems of soil degradation, particularly the soil fertility component, has become a major policy

issue in Ethiopia since the agricultural sector is the major pillar of economic and social development (Neglo et al. 2021; Haregeweyn et al. 2015).

The Suha watershed has a high potential for crop production in the northwestern highlands of Ethiopia. However, the natural resource base, particularly the soil resource, has been under stress due to high population pressure, and the soil condition has been deteriorating over time. The problems of soil fertility remain a major challenge and have not yet been reversed in this watershed (Simane et al. 2013). Maintaining soil quality is a key factor for agricultural productivity and environmental sustainability (Elias 2004). However, soil quality deterioration and nutrient depletion are major problems observed in the north western highlands of Ethiopia including the study watershed. Moreover, the alteration of soil quality indicators is site-specific due to the difference in soil management strategies, agroecologies, soil types and landscape positions. Therefore, regular monitoring of soil quality is vital to know whether this parameter is maintained or degraded and to apply appropriate and site-specific management strategies to improve soil conditions. However, the level of soil quality degradation was not well known and documented in the study area and the results of this study could provide new information to improve soil conditions. Hence, the results of this study could provide information to land managers and experts on how to select sets of site-specific soil management strategies to increase agricultural productivity. Therefore, the general objective of this study was to evaluate the status of soil quality in response to land use and cover change in terms of soil physical and chemical fertility along the topo-sequence of the Suha watershed, in the northwestern highlands of Ethiopia.

Methodology

Study area description

The study area, the Suha watershed, is located in the north-western highlands of Ethiopia and geographically lies from 37° 56' 15" to 38° 18' 49" E and from 10° 06' 46" to 10° 41' 56" N (Fig. 1). Its elevation ranges from 1040 to 3986 m above sea level and covers 80,343 ha. This watershed has seven topographic categories (from flat to very steep) based on FAO's (2006) classification.

Climate

The climate is the most important factor that determines the type of land use system and has a greater impact on agricultural productivity. The study area has a unimodal type of rainfall that extends from June to September, and based on the data obtained from nearby stations (Bichena, Dejen, Kuy, and Robgebeya),

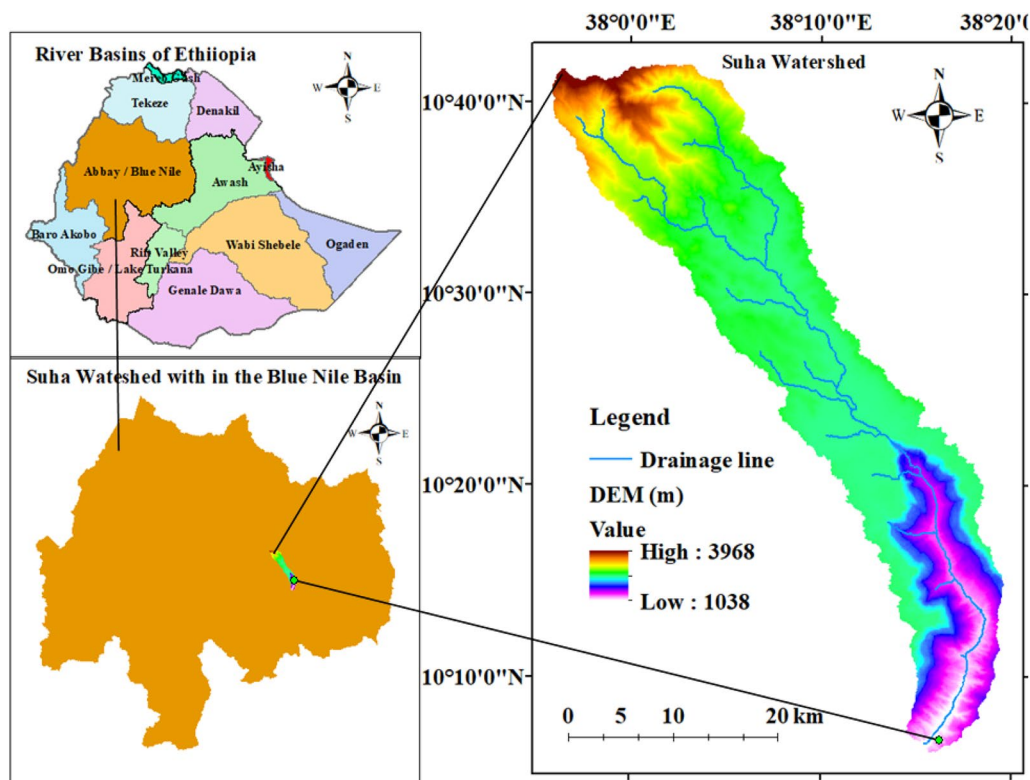


Fig. 1 Map of the study area

Table 1 Land use and cover types and their area coverage in the Suha watershed

Land use/cover	Area (ha)	Proportion in %
Agricultural land	59,731.5	74.4
Grazing land	7193.8	9.1
Forest land	1076.5	1.3
Shrub land	3897.1	4.9
Bare land	6714.9	8.4
Built up	1719.7	2.0
Total	80,333.5	100

it receives annual rainfall that ranges from 1213 mm at the lower part to 1396 mm in the upper part of the watershed. The mean minimum and mean maximum temperatures are 8.3 °C and 23.6 °C respectively.

According to (Yeneneh et al. 2022), six types of land use systems were identified in the study area; including agricultural land (74.4%), grazing land (9.1%), forest land (1.3%), shrub land (4.9%), bare land (8.4%), and built-up area (2%) (Table 1).

Site selection and soil sample collection

Before soil sample collection, the watershed was stratified into three strata based on its altitudinal gradient using the digital elevation model (DEM) of the watershed. These strata were classified as the upper (2700–3900 m.a.s.l.), middle (2400–2700 m.a.s.l.), and lower (1080–2400 m.a.s.l.) parts of the watershed. Then, tentative soil sampling sites were fixed on the land use and land cover maps along the top sequence of the watershed. In addition, a reconnaissance field survey and discussion with development agents were carried out to get a general overview of the watershed. A total of 27 composite surface soil samples (0–30 cm) were collected using a soil auger in three replications from adjacently located land use systems (agricultural land, grazing land, and forest land) having similar slope gradients and soil types in each stratum (upper, middle, and lower parts of the watershed) (Fig. 2). Ten to fifteen primary samples were collected to prepare one composite sample. Soil quality indicators including soil particle size distribution, ISS, soil water contents, soil pH-H₂O, organic carbon (OC), total nitrogen (TN), carbon to nitrogen ratio (C:N), available phosphorus (Av. P), Exchangeable bases (Na⁺, K⁺, Ca²⁺, and

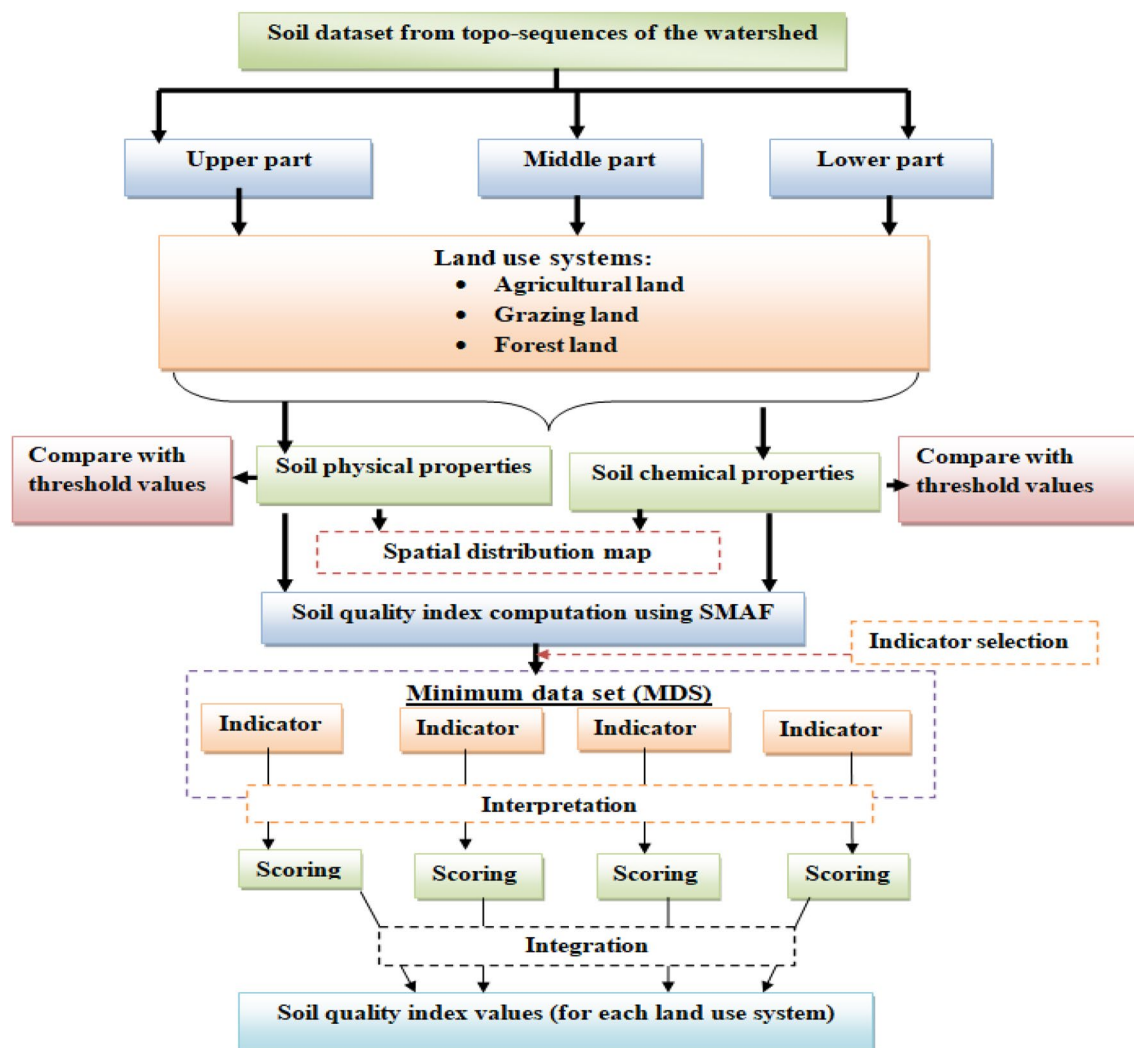


Fig. 2 The general framework of the methodology used

Mg²⁺), PBS, and cation exchange capacity (CEC) were considered in this study. Data on the geographical locations of the sampling sites were also gathered using a global positioning system (GPS).

Sample preparation and laboratory analysis

One kilogram of composite samples was prepared, packed in plastic bags, labeled, and taken to Ethiopia Water Works, Design, and Supervision Corporation for analysis. Soil samples were air-dried, crushed, and sieved through a 2 mm and 0.5 mm mesh sieve to remove coarser particles. Standard laboratory procedures were followed in the analysis of soil physical and chemical quality indicators.

Analysis of soil physical properties

Particle size distribution was determined by the hydrometric method (Bouycous 1962). The index of soil aggregate stability (ISS) was determined using Eq. (1) as described by (Pieri 1992). The soil water contents, FC and PWP, were determined at 1/3 bar and 15 bar, respectively, by the pressure membrane suction method (Estefan et al. 2013). Then, plant available water holding capacity (AWHC) was calculated using Eq. (2).

$$ISS = \frac{OC \times 1.724}{(L + A)} \times 100, \quad (1)$$

where ISS = index of soil aggregate stability; OC = organic carbon; L = silt proportion, and A = clay proportion.

$$AWHC (\%) = FC - PWP, \quad (2)$$

where AWHC is the available water holding capacity, FC is the field capacity; and PWP is the permanent wilting point.

Analysis of soil chemical properties

Soil pH was determined in water (pH-H₂O) using a 1:2.5 soil-to-water solution ratio with a pH meter, as outlined in Van Reeuwijk (2006). The Walkley and Black wet digestion method was used to analyze soil OC content (Van Ranst 1993). Total N was analyzed using the Kjeldahl digestion, distillation, and titration method as described by Bremner and Mulvaney (1982). Available soil P was analyzed according to the standard procedure of Olsen et al. (1954). Exchangeable bases (Ca²⁺, Mg²⁺, K⁺, and Na⁺) were determined by ammonium acetate (1N NH₄OAc) at pH 7.0. Monovalent (K⁺ and Na⁺) and divalent (Ca²⁺ and Mg²⁺) cations were determined by using a flame photometer and an atomic absorption spectrophotometer, respectively (van Ranst 1993). The CEC of the soil was determined from ammonium-saturated samples that were subsequently replaced by Na from a percolated sodium chloride solution. The excess salt was removed by washing with alcohol, and the ammonium that was displaced by sodium was measured using the Kjeldahl method (Chapman 1965).

Percent base saturation (PBS) was calculated by dividing the sum of basic cations (Ca²⁺, Mg²⁺, K⁺, and Na⁺) by the CEC of the soil (Eq. 3).

$$BS\% = \frac{Ca^{2+} + Mg^{2+} + K^{+} + Na^{+} \text{ (cmol(+) / kg)}}{CEC \text{ (cmol(+) / kg)}} \times 100, \quad (3)$$

where BS% is base saturation in percent (PBS).

$$ESP = \frac{Na^{+} \text{ (cmol(+) / kg)}}{CEC \text{ (cmol(+) / kg)}} \times 100, \quad (4)$$

where ESP is the exchangeable sodium percentage.

Soil quality deterioration index (SQDI) for each soil quality indicator

The soil quality deterioration index (SQDI) for each soil quality indicator under different land use systems and elevation gradients was computed using an undisturbed ecosystem (forest land) as a reference and evaluated the soil parameters of other land systems against this reference as described by (Abera and Assen 2019; Gui et al. 2009). Deterioration index (DI) Values were computed by taking the difference between the mean values of quality indicators in the given land use and the mean values in the referenced land use system. From the total

of 19 soil quality indicators, 13 indicators (clay, silt, ISS, AWC, pH, OC, TN, Av. P, Ca²⁺, Mg²⁺, Na⁺, K⁺, and CEC) were selected using an expert opinion (Anderson 2002) to compute SQDI values. Most of these indicators are sensitive to land use change and land management (Gui et al. 2009; Abera and Assen 2019).

Soil quality index (SQI) computation: specific to production and soil erosion susceptibility

Individual soil quality indicators wouldn't give a proper estimation of soil quality, and hence a combination of soil quality indicators is required. So far, different approaches have been developed to assess soil quality. However, there is no universally accepted technique that could be applied to evaluate soil conditions in various ecosystems (Mukherjee and Lal 2014; De Paul Obade and Lal 2016). The model developed by Bajracharya et al. (2007) and also used by other researchers (Feleke et al. 2019) was adopted for this study. In this approach, weighting factors and ratings of selected soil quality indicators were used to develop soil quality ratings (SQR). The ratings for these parameters were adopted from EthioSIS (2014). The model has the following form:

$$SQR = [(a * RSTC) + (b * RpH) + (c * ROM) + (d * RNPK)], \quad (5)$$

where a, b, c, and d are weighting values for the four soil properties. RSTC=ranking value for soil texture; RpH=value for soil pH; ROM=ranking value for soil organic matter; and RNPK=value for soil nutrient contents of nitrogen, phosphorus, and potassium. These soil parameters are commonly used to evaluate the soil quality index for a given land use system or ecosystem.

Statistical analysis

A two-way analysis of variance (ANOVA) following the generalized linear model (GLM) procedure was applied to test the difference between the mean values of soil quality indicators in different land use systems, the elevation gradient, and their interaction effects. The mean values of statistically different soil quality indicators were separated by Duncan's Multiple Range Test using SAS software version 9.0. In addition, linear correlation analysis was performed to explore the relationships among soil properties. The study area and soil maps were prepared using ArcGIS software version 9.5.

Results

The results of descriptive statistics illustrated the minimum and maximum values of soil quality indicators which revealed the existence of a high degree of variability in the study watershed (Table 2). The differences

Table 2 Descriptive statistics of the analyzed soil physicochemical properties in the Suha Watershed

Soil properties	No. of samples	Minimum	Maximum	Mean	SE	CV (%)
Physical properties						
Sand (%)	27	0.24	49.06	13.21	2.57	177.98
Clay (%)	27	5.66	86.04	53.72	3.73	375.72
Silt (%)	27	12.47	45.28	33.07	1.77	84.9
ISS	27	1.05	10.78	4.65	0.53	7.60
FC (%)	27	36.47	72.41	48.57	1.59	68.70
PWP (%)	27	25.12	49.28	33.98	1.11	33.41
AWHC (%)	27	10.00	23.13	14.59	0.57	8.73
Chemical properties						
pH-H ₂ O	27	4.93	6.31	5.78	0.06	0.11
OC (%)	27	0.38	4.60	2.27	0.23	1.44
TN (%)	27	0.05	0.35	0.15	0.01	0.00
C:N	27	7.6	20.14	15.11	0.72	13.83
Av.P (mg kg ⁻¹)	27	10.10	64.50	27.20	3.87	223.27
Ex. Na ⁺ (cmol(+) kg ⁻¹)	27	0.48	2.51	1.37	0.09	0.21
Ex. K ⁺ (cmol(+) kg ⁻¹)	27	0.17	3.28	1.07	0.13	0.49
Ex. Ca ²⁺ (cmol(+) kg ⁻¹)	27	18.94	58.77	37.69	2.07	116.26
Ex. Mg ²⁺ (cmol(+) kg ⁻¹)	27	7.36	22.14	13.59	0.78	16.74
CEC (cmol(+) kg ⁻¹)	27	45.58	88.55	69.44	2.24	135.31
BS (%)	27	56.9	99.4	76.48	2.24	148.51

in the land use systems and elevation gradient might be the factors contributing to this variability. The results of ANOVA, correlation analysis, and PCA are presented and discussed sequentially.

Effects of land use systems and elevation gradients on selected soil physical quality indicators

The analysis of variance results of soil physical quality indicators under different land use types and elevation gradients were depicted in (Table 3). The results revealed that soil particles (sand, silt, and clay) showed significant differences ($p < 0.05$) between agricultural and forest land use types in the upper part of the watershed. In the other cases, non-significant differences were observed in the main and interaction effects of land use systems and locations. The mean value of sand ranged from 2.55 to 35.12%; for the clay fraction, it ranged from 19.85 to 72.66%; and for the silt fraction, it ranged from 24.78 to 47.36%. The textural class of soil in most locations and land use systems is clay; other textural classes include clay loam, silt clay, and sand clay. When the clay fraction is removed from the surface by soil erosion, the content of sand increases, as evidenced by the negative and significant correlation ($r = -0.755$, $p < 0.01$) between these fractions.

Significant differences ($p < 0.05$) were also observed for the Index of Soil Aggregate Stability (ISS) among

treatments (land use systems and elevation gradients) and their interaction effects. The lowest (2.06%) and highest (7.21%) values were recorded in the agricultural land and forest land use systems, respectively. Pieri (1992) classified structural degradation of soils as stable structure when $ISS > 9\%$; low risk of structural degradation when $7\% < ISS < 9\%$; high risk of structural degradation when $5\% < ISS < 7\%$; and structurally degraded soil when $ISS < 5\%$. Based on these classifications, soils under agricultural and grazing lands are rated as structurally degraded soils, whereas soils under forest land are rated as having a low risk of structural degradation.

The mean value of soil water content at field capacity (FC) ranged from 36.94 to 59.08% in the clay loam textural class and from 40.31 to 72.41% for the clay textural class, the values of PWP ranged from 25.52 to 43.91% and from 28.84 to 49.28% for the same textural classes. For the above textural classes, the values of AWC ranged from 11.42 to 15.17%; and from 11.47 to 23.13% respectively. Significant differences were observed between the mean values of PWP in the agricultural land and forest land in the midland of the watershed. Though there is no statistically significant difference among most of the soil physical quality indicators, marked differences were observed as they are altered by land use systems and elevation gradients.

Table 3 Mean values of sand, clay, silt, ISS, FC, PWP, and AWC under different land use systems and elevation gradients

Soil property	Elevation gradient	Land use system			Mean $\pm$ SE
		Cultivated land	Grazing land	Forest land	
Sand	Upper	2.55b $\pm$ 6.6	15.18ab $\pm$ 6.6	35.12a $\pm$ 6.6	17.62 $\pm$ 3.84
	Middle	14.78a $\pm$ 6.6	14.34a $\pm$ 6.6	3.19a $\pm$ 6.6	10.76 $\pm$ 3.84
	Lower	15.45a $\pm$ 6.6	7.62a $\pm$ 6.6	10.65a $\pm$ 6.6	11.24 $\pm$ 3.84
	Mean $\pm$ SE	10.93a $\pm$ 3.84	12.38a $\pm$ 3.84	16.32a $\pm$ 3.84	13.21 $\pm$ 2.22
Clay	Upper	72.66a $\pm$ 9.4	49.94a $\pm$ 9.4	19.85a $\pm$ 9.4	47.48 $\pm$ 5.45
	Middle	54.21a $\pm$ 9.4	51.53a $\pm$ 9.4	61.03a $\pm$ 9.4	55.59 $\pm$ 5.45
	Lower	56.53a $\pm$ 9.4	56.52a $\pm$ 9.4	61.20a $\pm$ 9.4	58.08 $\pm$ 5.45
	Mean $\pm$ SE	61.13a $\pm$ 5.45	52.66a $\pm$ 5.45	47.36a $\pm$ 5.45	53.72 $\pm$ 3.15
Silt	Upper	24.78b $\pm$ 5.0	34.88ab $\pm$ 5.0	45.03a $\pm$ 5.0	34.89 $\pm$ 2.89
	Middle	31.02a $\pm$ 5.0	34.13a $\pm$ 5.0	35.77a $\pm$ 5.0	33.64 $\pm$ 2.89
	Lower	28.03a $\pm$ 5.0	35.86a $\pm$ 5.0	27.14a $\pm$ 5.0	30.68 $\pm$ 2.89
	Mean $\pm$ SE	27.94a $\pm$ 2.89	34.96a $\pm$ 2.89	36.31a $\pm$ 2.89	33.07 $\pm$ 1.67
Aggregate stability	Upper	1.78c $\pm$ 0.78	4.12b $\pm$ 0.78	10.29a $\pm$ 0.78	5.40a $\pm$ 0.46
	Middle	2.32b $\pm$ 0.78	4.95a $\pm$ 0.78	5.42a $\pm$ 0.78	4.23a $\pm$ 0.46
	Lower	2.09b $\pm$ 0.78	4.95a $\pm$ 0.78	5.92a $\pm$ 0.78	4.32a $\pm$ 0.46
	Mean $\pm$ SE	2.05c $\pm$ 0.46	4.67b $\pm$ 0.46	7.21a $\pm$ 0.46	4.65 $\pm$ 0.26
FC (v %)	Upper	48.97a $\pm$ 4.36	50.71a $\pm$ 4.36	41.76a $\pm$ 4.36	47.14 $\pm$ 2.52
	Middle	41.24a $\pm$ 4.36	46.62a $\pm$ 4.36	48.63a $\pm$ 4.36	45.49 $\pm$ 2.52
	Lower	60.10a $\pm$ 4.36	47.35a $\pm$ 4.36	51.71a $\pm$ 4.36	53.05 $\pm$ 2.52
	Mean $\pm$ SE	50.1a $\pm$ 2.52	48.23a $\pm$ 2.52	47.37a $\pm$ 2.52	48.56 $\pm$ 1.45
PWP (v %)	Upper	33.59a $\pm$ 2.95	35.2a $\pm$ 2.95	29.09a $\pm$ 2.95	32.62 $\pm$ 1.71
	Middle	28.31b $\pm$ 2.95	32.57ab $\pm$ 2.95	34.79a $\pm$ 2.95	31.89 $\pm$ 1.71
	Lower	42.21a $\pm$ 2.95	34.04a $\pm$ 2.95	36.00a $\pm$ 2.95	37.41 $\pm$ 1.71
	Mean $\pm$ SE	34.71a $\pm$ 1.71	33.94a $\pm$ 1.71	33.29a $\pm$ 1.71	33.98 $\pm$ 0.99
AWC (v %)	Upper	15.39a $\pm$ 1.71	15.51a $\pm$ 1.71	12.67a $\pm$ 1.71	14.52 $\pm$ 0.99
	Middle	12.92a $\pm$ 1.71	14.05a $\pm$ 1.71	13.84a $\pm$ 1.71	13.60 $\pm$ 0.99
	Lower	17.89a $\pm$ 1.71	13.31a $\pm$ 1.71	15.71a $\pm$ 1.71	15.63 $\pm$ 0.99
	Mean $\pm$ SE	15.40a $\pm$ 0.99	14.29a $\pm$ 0.99	14.07a $\pm$ 0.99	14.59 $\pm$ 0.57

The mean values for the same letter are not significant

Effects of land use systems and elevation gradients on selected chemical soil quality indicators

Soil pH, OC, TN, C:N, and available phosphorus (Av. P)

The analysis of variance results revealed that the mean values of soil pH ranged from 5.56 to 6.13 (Table 4), which is classified as moderately acidic (EthioSIS 2014). The lowest value (5.56) and the highest value (6.13) were found in the mid-altitude and low altitudes of the cultivated fields. Non-significant variations were observed between land use systems, elevation gradients, and their interaction effects in the upper and mid altitudes; whereas, in the lower altitude, significant variations were observed between cultivated land and other land use systems.

The mean value of soil organic carbon (SOC) ranged from 1.01 to 3.89%. In all elevation gradients, the lowest values were recorded in the agricultural land, whereas the highest values were in the forest land (Table 4).

Significant differences were observed between the main treatments (land use types and locations) and their interaction effects (LU*L). Based on the ratings of EthioSIS (2014), the contents of SOC are very low in the agricultural lands in all locations, low in the grazing lands, and moderate in the forest land use system. This indicates that soil quality, particularly soil biological indicators, is highly degraded in the cultivated and grazing land use systems (Fig. 3).

The results also showed that the mean values of soil TN ranged from 0.08% (agricultural land) to 0.21% (forest land) (Table 4). Significant differences ($p < 0.05$) were observed in the main and interaction effects of land use systems and elevation gradients. A significant difference was observed between agricultural land and the other two land use systems. Based on the ratings of EthioSIS (2014), the status of TN is rated as very low in agricultural land and moderate in forest

Table 4 Mean values of pH, OC, TN, C:N, and Av. P under different land use systems and elevation gradients

Soil property	Elevation gradient	Land use system			Mean $\pm$ SE
		Cultivated land	Grazing land	Forest land	
pH-H ₂ O	Upper	5.85a $\pm$ 0.19	5.64a $\pm$ 0.19	5.99a $\pm$ 0.19	5.82 $\pm$ 0.11
	Middle	5.56a $\pm$ 0.19	5.61a $\pm$ 0.19	5.83a $\pm$ 0.19	5.67 $\pm$ 0.11
	Lower	6.13a $\pm$ 0.19	5.62b $\pm$ 0.19	5.78b $\pm$ 0.19	5.84 $\pm$ 0.11
	Mean $\pm$ SE	5.84a $\pm$ 0.11	5.69a $\pm$ 0.11	5.79a $\pm$ 0.11	5.78 $\pm$ 0.07
OC	Upper	1.01b $\pm$ 0.48	2.05b $\pm$ 0.48	3.89a $\pm$ 0.48	2.32 $\pm$ 0.28
	Middle	1.12b $\pm$ 0.48	2.49ab $\pm$ 0.48	3.04a $\pm$ 0.48	2.22 $\pm$ 0.28
	Lower	1.09b $\pm$ 0.48	2.69ab $\pm$ 0.48	3.07a $\pm$ 0.48	2.29 $\pm$ 0.28
	Mean $\pm$ SE	1.07c $\pm$ 0.28	2.41b $\pm$ 0.28	3.33a $\pm$ 0.28	2.27 $\pm$ 0.16
TN	Upper	0.09b $\pm$ 0.03	0.13b $\pm$ 0.03	0.21a $\pm$ 0.03	0.14 $\pm$ 0.02
	Middle	0.08a $\pm$ 0.03	0.17a $\pm$ 0.03	0.18a $\pm$ 0.03	0.14 $\pm$ 0.02
	Lower	0.08b $\pm$ 0.03	0.18b $\pm$ 0.03	0.21a $\pm$ 0.03	0.16 $\pm$ 0.02
	Mean $\pm$ SE	0.08b $\pm$ 0.02	0.17a $\pm$ 0.02	0.20a $\pm$ 0.02	0.15 $\pm$ 0.01
C:N	Upper	11.22b $\pm$ 2.02	15.89a $\pm$ 2.02	18.71a $\pm$ 2.02	15.27 $\pm$ 1.17
	Middle	14.38a $\pm$ 2.02	13.83a $\pm$ 2.02	18.57a $\pm$ 2.02	15.59 $\pm$ 1.17
	Lower	13.36a $\pm$ 2.02	14.59a $\pm$ 2.02	15.04a $\pm$ 2.02	14.44 $\pm$ 1.17
	Mean $\pm$ SE	12.98b $\pm$ 1.17	14.77ab $\pm$ 1.17	17.56a $\pm$ 1.17	15.11 $\pm$ 0.67
Av.P	Upper	45.00a $\pm$ 4.73	18.63b $\pm$ 4.73	27.80ab $\pm$ 4.73	30.47 $\pm$ 2.73
	Middle	27.83a $\pm$ 4.73	14.93a $\pm$ 4.73	13.10a $\pm$ 4.73	18.62 $\pm$ 2.73
	Lower	54.17a $\pm$ 4.73	20.30b $\pm$ 4.73	23.03b $\pm$ 4.73	32.5 $\pm$ 2.73
	Mean $\pm$ SE	42.33a $\pm$ 2.73	17.95b $\pm$ 2.73	21.31b $\pm$ 2.73	27.2 $\pm$ 1.58

The means in the rows followed by the same letter are not significant

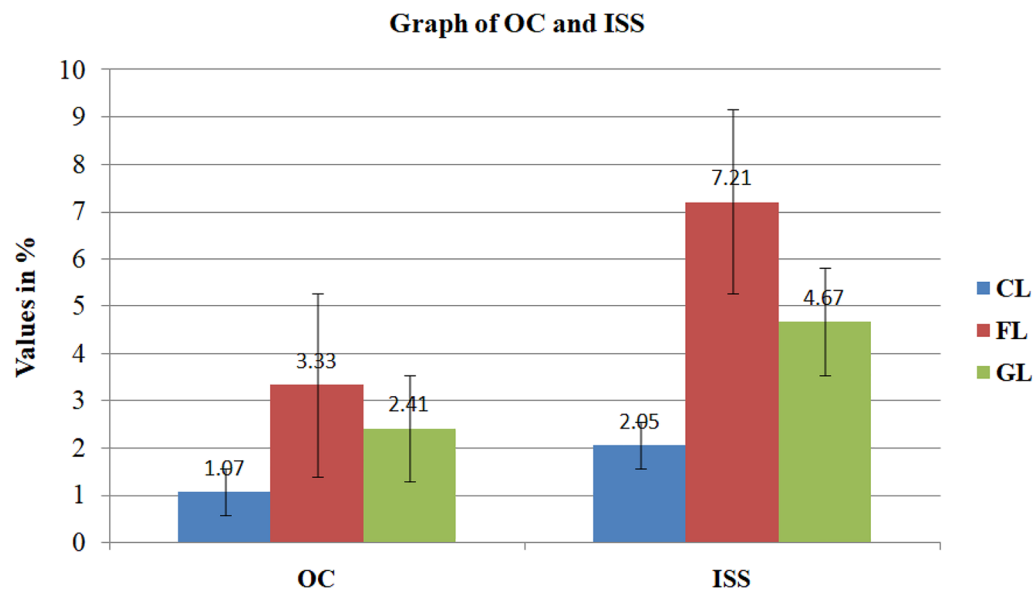


Fig. 3 The relationship between OC and ISS under different land use systems. OC organic carbon, ISS index of soil aggregate stability, CL cultivated land, GL grazing land, FL forest land

and grazing land use systems. The mean values of the C:N ratio ranged from 11.22 to 18.71, and the overall mean values were 12.98, 14.77, and 17.5 in the cultivated land, grazing land, and forest land, respectively. A significant difference was observed between cultivated land and forest land. Though non-significant variations were observed, the numerical values were higher on the grazing land as compared to the cultivated land. The same is true for forest land and grazing land (Table 4).

The value of available phosphorus was significantly higher (42.33 mg kg^{-1}) in the cultivated land, followed by the forest land (21.31 mg kg^{-1}) and the grazing land (17.95 mg kg^{-1}). Significant differences ($p < 0.05$) were also observed in the mean values among locations, where the higher value was observed in the lower part of the watershed and the lowest value in the

middle part where there is Vertisol coverage (Table 4). The status of available phosphorus is low in the grazing and forest land use systems and moderate in the agricultural land, whereas the overall mean value (27.2 mg kg^{-1}) was rated as low (EthioSIS 2014).

Exchangeable bases (Na^+ , K^+ , Ca^{2+} , and Mg^{2+}), CEC, and PBS

There is no significant difference in all exchangeable bases in the main and interaction effects of land use systems and elevation gradients (Table 5). However, marked variations were observed among land use systems. In all locations, the lowest values of exchangeable Na^+ and K^+ were found in the cultivated land, and the highest values in the forest land. The same is true for exchangeable Ca^{2+} and Mg^{2+} . Since the critical value of exchangeable Na^+ varied due to the variation in CEC, clay content, and type of crop grown, its

Table 5 Mean values of exchangeable bases (Na^+ , K^+ , Ca^{2+} , and Mg^{2+}), CEC, and PBS under different land use systems and elevation gradients

Soil property	Elevation gradient	Land use system			Mean $\pm$ SE
		Cultivated land	Grazing land	Forest land	
Ex. Na	Upper	1.32a $\pm$ 0.25	1.22a $\pm$ 0.25	1.21a $\pm$ 0.25	1.25 $\pm$ 0.14
	Middle	1.51a $\pm$ 0.25	1.20a $\pm$ 0.25	1.52a $\pm$ 0.25	1.41 $\pm$ 0.14
	Lower	0.94a $\pm$ 0.25	1.44a $\pm$ 0.25	1.98a $\pm$ 0.25	1.46 $\pm$ 0.14
	Mean $\pm$ SE	1.25a $\pm$ 0.14	1.29a $\pm$ 0.14	1.57a $\pm$ 0.14	1.37 $\pm$ 0.08
Ex. K	Upper	0.84b $\pm$ 0.38	0.55b $\pm$ 0.38	2.00a $\pm$ 0.38	1.13 $\pm$ 0.22
	Middle	0.84a $\pm$ 0.38	1.65a $\pm$ 0.38	0.97a $\pm$ 0.38	1.15 $\pm$ 0.22
	Lower	0.90a $\pm$ 0.38	0.76a $\pm$ 0.38	1.12a $\pm$ 0.38	0.93 $\pm$ 0.22
	Mean $\pm$ SE	0.86a $\pm$ 0.22	0.99a $\pm$ 0.22	1.36a $\pm$ 0.22	1.07 $\pm$ 0.13
Ex. Ca	Upper	33.32a $\pm$ 6.80	42.10a $\pm$ 6.80	39.04a $\pm$ 6.80	38.15 $\pm$ 3.93
	Middle	28.82a $\pm$ 6.80	33.45a $\pm$ 6.80	39.90a $\pm$ 6.80	34.09 $\pm$ 3.93
	Lower	51.97a $\pm$ 6.80	40.00a $\pm$ 6.80	41.2a $\pm$ 6.80	40.85 $\pm$ 3.93
	Mean $\pm$ SE	34.48a $\pm$ 3.93	38.53a $\pm$ 3.93	40.07a $\pm$ 3.93	37.70 $\pm$ 2.27
Ex. Mg	Upper	13.28a $\pm$ 2.53	15.93a $\pm$ 2.53	11.05a $\pm$ 2.53	13.42 $\pm$ 1.46
	Middle	10.41a $\pm$ 2.53	13.87a $\pm$ 2.53	14.40a $\pm$ 2.53	12.89 $\pm$ 1.46
	Lower	21.42a $\pm$ 2.53	13.96a $\pm$ 2.53	13.21a $\pm$ 2.53	14.45 $\pm$ 1.46
	Mean $\pm$ SE	13.30a $\pm$ 1.46	14.59a $\pm$ 1.46	12.89a $\pm$ 1.46	13.59 $\pm$ 0.84
CEC	Upper	63.95a $\pm$ 7.00	69.86a $\pm$ 7.00	69.10a $\pm$ 7.00	67.63 $\pm$ 4.04
	Middle	59.62b $\pm$ 7.00	66.17ab $\pm$ 7.00	72.71a $\pm$ 7.00	66.17 $\pm$ 4.04
	Lower	74.04a $\pm$ 7.00	69.08a $\pm$ 7.00	80.43a $\pm$ 7.00	74.52 $\pm$ 4.04
	Mean $\pm$ SE	65.87a $\pm$ 4.04	68.37a $\pm$ 4.04	74.07a $\pm$ 4.04	69.44 $\pm$ 2.33
PBS	Upper	74.97a $\pm$ 8.11	82.23a $\pm$ 8.11	77.00a $\pm$ 8.11	78.06 $\pm$ 4.68
	Middle	70.23a $\pm$ 8.11	75.17a $\pm$ 8.11	78.43a $\pm$ 8.11	74.61 $\pm$ 4.68
	Lower	78.03a $\pm$ 8.11	79.27a $\pm$ 8.11	72.93a $\pm$ 8.11	76.76 $\pm$ 4.68
	Mean $\pm$ SE	74.13a $\pm$ 4.68	78.89a $\pm$ 4.68	86.12a $\pm$ 4.68	76.48 $\pm$ 2.70
ESP	Upper	2.22 $\pm$ 0.43	1.84 $\pm$ 0.43	1.73 $\pm$ 0.43	1.93 $\pm$ 0.25
	Middle	2.63 $\pm$ 0.43	1.86 $\pm$ 0.43	2.09 $\pm$ 0.43	2.19 $\pm$ 0.25
	Lower	1.21 $\pm$ 0.43	2.11 $\pm$ 0.43	2.54 $\pm$ 0.43	1.95 $\pm$ 0.25
	Mean $\pm$ SE	2.02 $\pm$ 0.25	1.94 $\pm$ 0.25	2.12 $\pm$ 0.25	2.03 $\pm$ 0.14

Means in the same row followed by the same letter are not significant

threshold value was more explained by ESP, in which an ESP value of 15% is critical for most crops (Anon 1954). The lowest (1.92%) and highest (2.12%) values of ESP were observed in the grazing and forest land use systems, respectively. The soils of the study area are free of sodicity problems as the values of ESP are far less than 15%. Based on the classifications of EthioSIS (2014), the contents of exchangeable bases are rated as high in all locations and land use systems.

The mean values of CEC didn't show significant differences between the main treatments (land use type and locations) and their interaction effects (LU*L), except in the midland of the watershed, where a significant difference was observed between cultivated land and forest land (Table 5). Even though non-significant differences were found in the other cases, the numerical values substantially varied between land use types and locations. The highest overall mean value (74.07 cmol(+) kg⁻¹) was found under the forest land, followed by grazing land (68.37 cmol(+) kg⁻¹) and the lowest value (65.69 cmol(+) kg⁻¹) under the cultivated land (Table 5). Based on Landon's (2014) classifications, the contents of CEC are rated as very high in all

land use systems and locations, indicating that soils are inherently fertile due to the parent materials from which they are developed (Elias 2019; William 2016). The highest mean value (86.12%) of PBS was observed in the forest land, followed by grazing land (78.89%), and the lowest value in the cultivated field (74.13%). These values are rated as high and are indicators of good soil fertility.

Soil quality deterioration index (SQDI) for each quality indicator

Soil quality deterioration index results revealed that most of the soil quality indicators have negative values in the cultivated and grazing lands, indicating that the soil condition is becoming poor due to poor management in these land use systems. In the cultivated field, the highest negative index value was for ISS (−71.3%), followed by SOC (−67.7%). On the other hand, the highest positive index value for available P (98.6%) was in the cultivated land. For grazing land, the highest negative values were recorded for ISS (−35.1%) and SOC (−27.7%) (Fig. 4). The total SQDI values of cultivated and grazing lands were −12.34% and −10.0, respectively.

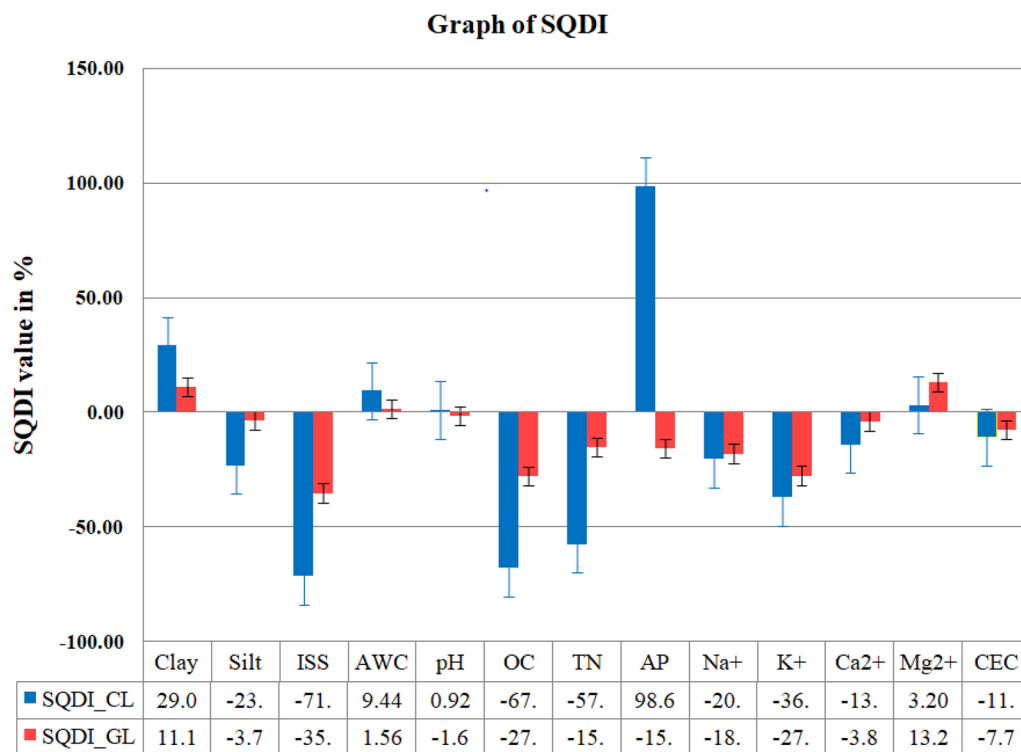


Fig. 4 SQDI values of soil quality indicators under cultivated and grazing lands based on the referenced undisturbed ecosystem (forest land). SQDI soil quality deterioration index, CL cultivated land, GL grazing land

Table 6 Status of soil quality under different land use systems based on defined ratings (Li et al. 2018)

Land use system	SQI values for selected indicators				Total SQI	Rating
	STC	pH	OC	NPK		
CL	0.04	0.06	0.16	0.12	0.38	Low
FL	0.04	0.06	0.32	0.18	0.60	High
GL	0.04	0.06	0.24	0.18	0.52	Moderate

CL cultivated land, FL forest land, GL grazing land, SQI soil quality index, STC soil texture, OC organic carbon, NPK nitrogen, phosphorus, and potassium

Soil quality index (SQI) computation from production and soil erosion perspectives

The overall SQI mean value under the cultivated land use system was significantly lower than the forest and grazing lands. The mean values were 0.38, 0.60, and 0.52 for cultivated land, forest land, and grazing land, respectively (Table 6). Based on Li et al.'s (2018) classification, the soil quality was rated as low, moderate, and high for cultivated land, grazing land, and forest land respectively.

Discussion

Effects of land use types and elevation gradients on selected physical soil quality indicators

The relative proportion of separate soil groups determines the soil textural class, which significantly affects other soil properties (physical, chemical, and biological properties). The silt fraction is the most mobile particle due to water erosion, and the proportions of these fractions are lower in poorly managed and intensively cultivated fields. In the current study, the lowest value of silt fraction was found in the cultivated field, followed by grazing land, which implies the problems of soil erosion in these land use systems. On the other hand, the highest value of clay fraction in the cultivated fields might be attributed to accelerated weathering because of soil disturbance.

Soil texture is an inherent property of soil and is the result of soil-forming processes. Its variation could be due to the illuviation of clay particles, the removal of clay by soil erosion, biological activities, upward movement of the sand fraction by swelling and shrinking phenomena, or combinations of these two or more processes (IUSS Working Group 2006). Even though it is less influenced by anthropogenic activities in a short period (Brejda et al. 2000), in the highland region where there is high rainfall and poor land use systems, smaller particle sizes will be eroded and transported to other locations, resulting in variations in the relative proportions of soil particles and textural classes. This illustrates the non-significant differences in soil particle sizes between land-use systems in the study area. This result agreed with some previous research findings. For instance, Mulat et al. (2021) demonstrated that the distribution of

soil in separate groups didn't show significant differences among land use systems in the Kersa watershed, Oromia region. Teferi et al. (2016) also reported similar findings from the Jedeb watershed in northwestern Ethiopia. Soil textural class significantly determines the content of soil water, its retention, and the availability of soil nutrients. For instance, the presence of a higher proportion of sand fraction will negatively affect water-holding capacity and nutrient availability (Abera and Assen 2019; Warra et al. 2015).

Soil aggregate stability is considerably influenced by land use systems and land management strategies. In this study, it was negatively affected in cultivated fields compared to soils under other land use systems. This could be attributed to the depletion of soil organic matter in the cultivated fields and the better accumulation of SOM in the forest and grazing lands. The findings of Delelegn et al. (2017) also explained that the value of soil aggregate stability in cultivated land is significantly lower than in other land use systems. Wei et al. (2014) also emphasized that intensive cultivation significantly decreases soil aggregate stability, whereas, in non-tilled landscapes (forest lands), macro-aggregates of soils are enhanced. Soil organic matter is a key factor in binding soil particles into aggregates. Due to tillage operations, soil organic matter could be further depleted, and hence macro aggregates will be broken down in the cultivated fields. The presence of an adequate amount of SOM in the soil enhances soil microbial and fungal biomass, which will improve soil aggregate stability (Wu et al. 2015). Mulat et al. (2021) reported higher soil aggregate stability in grazing land use as compared to fallow land and cultivated fields. Peng et al. (2016) also explained the role of soil organic matter in maintaining soil aggregate stability. Similarly, Gebreyesus et al. (2014) examined the negative impacts of soil organic carbon depletion on soil aggregate stability and its consequences for soil degradation. Devine et al. (2014) emphasized that undisturbed ecologies like soils under forest land have better aggregate stability than disturbed ecologies.

Exploring the status of soil water content is essential to devising an irrigation schedule and improving water use efficiency depending on the water requirements of plants.

Table 7 Pearson's correlation matrix of selected physical and chemical soil properties

	Sand	Clay	Silt	FC	PWP	AWC	pH	OC	TN	C:N	Av.P	Na	K	Ca	Mg	CEC	PBS
Sand	1	-0.755**	0.455**	-0.262	-0.261	-0.223	-0.006	0.029	-0.075	0.087	0.118	-0.182	0.214	-0.328*	-0.364*	-0.224	-0.373*
Clay		1	-0.705**	0.441*	0.932**	0.470**	0.210	-0.096	-0.066	0.076	-0.053	0.173	-0.147	0.492**	0.542**	0.548**	0.360*
Silt			1	-0.577**	-0.513**	-0.614**	-0.314	0.455**	0.459**	0.022	-0.330*	0.191	0.293	-0.249	-0.487**	-0.453**	-0.093
FC				1	0.975**	0.899**	0.524**	-0.023	-0.003	0.035	0.308	0.046	0.056	0.650**	0.764**	0.656**	0.557**
PWP					1	0.778**	0.527**	-0.005	-0.039	-0.045	0.292	0.003	0.024	0.650**	0.748**	0.611**	0.509**
AWHC						1	0.441*	-0.056	-0.085	0.185	0.294	0.122	0.111	0.685**	0.764**	0.646**	0.566**
pH							1	0.116	0.060	-0.028	0.338*	-0.512**	0.491**	0.486**	0.490**	0.456**	0.425*
OC								1	0.908**	0.487**	-0.503**	0.154	0.645**	0.459**	0.060	0.311	0.399*
TN									1	0.125	-0.519**	0.232	0.625**	0.419*	0.069	0.208	0.451**
C:N										1	-0.232	0.128	0.143	0.337*	0.169	0.417*	0.126
Av.P											1	-0.294	-0.186	-0.057	0.076	0.073	-0.183
Ex. Na												1	-0.227	0.082	-0.011	0.149	0.011
Ex. K													1	0.333*	0.114	0.132	0.463**
Ex. Ca														1	0.797**	0.834**	0.831**
Ex. Mg															1	0.695**	0.753**
CEC																1	0.425*
PBS																	1

**Significant at $p < 0.01$; *significant at $p < 0.05$

In the present study, the highest amounts of soil water contents (FC and PWP) were obtained in the cultivated field, and the lowest values were obtained in the forest land with different land use systems and elevation gradients. This variation could be attributed to the contents of the clay fraction and soil organic matter (SOM). This is also evidenced by the positive and significant correlation of clay with FC ($r=0.441$, $p<0.05$) and PWP ($r=0.932$, $p<0.01$) (Table 7). The variation in the contents of available water holding capacity (AWHC) among different land use systems could be due to the difference in the contents of clay fraction and soil organic matter (Alawamy et al. 2021).

Effects of land use systems and elevation gradients on selected chemical soil quality indicators

Soil pH, SOC, TN, C:N, and available phosphorus (Av. P)

Even though the values of soil pH didn't show significant differences among land use systems except at location 3 (lower elevation), the soil condition is moderately acidic in all land use systems and locations. Even though the study area is found in the northwestern highlands of Ethiopia, where there is high rainfall and rocky topography, the problem of soil acidity is not critical. This might be attributed to the content of SOM and high CEC, in which basic cations are adsorbed on the surfaces of these colloidal particles. This is also evidenced by positive and significant correlations of pH with CEC ($r=0.456$, $p<0.01$), PBS ($r=0.425$, $p<0.05$), exchangeable Ca^{2+} ($r=0.486$, $p<0.01$), exchangeable K^+ ($r=0.491$, $p<0.01$), and exchangeable Mg^{2+} ($r=0.490$, $p<0.01$) (Table 7). On the other hand, the lowest values of soil pH in some locations and land uses could be due to the removal of basic cations through soil erosion, leaching, application of acid-forming fertilizers, crop harvest, and residue in the cultivated fields, and leaching and removal through soil erosion in the forest and cultivated lands, respectively. This result is in agreement with the findings of Teferi et al. (2016), who reported a non-significant difference among different land use systems in the Jedeb watershed, in the northwestern highlands of Ethiopia. On the contrary, Mulat et al. (2021) found significant differences among land use systems in the Kersa watershed, Oromia region. These differences could be due to differences in soil types, climatic conditions, and management strategies. Soil pH indicates the toxicity level of aluminum (Al) and the status (deficiency) of soil micronutrients.

Soil organic matter (SOC) is an important soil parameter that highly influences soil's physical, chemical, and biological properties. It is the key indicator of soil quality and is significantly influenced by anthropogenic activities, vegetation cover, and climatic conditions (Yu et al. 2020). In this study, the lowest values were found in

cultivated lands in all elevation gradients, and the values are far below the critical level. The content is also low in the grazing land but optimal in the forest land. Further decomposition of SOM due to continuous and intensive cultivation in the cultivated fields, depletion through soil erosion, complete removal of crop residues from the field (sources of SOM), and low input of organic fertilizers (crop residues and manure) are the major factors for extensive depletion of SOM in the cultivated fields. This leads to a high loss of soil nutrients from these fields, as the SOM is the most important soil parameter that determines soil fertility. On the other hand, low soil erosion, continuous litter fall to the soil, and the microclimatic conditions under the forest land (low organic matter decomposition) contribute to low soil organic carbon loss and the buildup of SOM in the forest land. These results agreed with the findings of Mulat et al. (2021), who reported a higher content of SOM in the grazing land than in cultivated and fallow lands in the Kersa watershed in eastern and Northern Ethiopia. Feleke et al. (2019) and Ryan et al. (2018) also demonstrated a higher content of SOM in protected grassland than in open grassland due to the frequent turnover of grass and dense root biomass in the protected land use. Similarly, Elias (2019) emphasized the negative impacts of the complete removal of crop residues from cultivated fields and using cow dung as a source of fuel instead of as soil amendments on the loss of soil nutrients and SOM.

Land use and land management strategies significantly affect the content of soil TN in the study area. In poorly managed cultivated fields, the content of TN is significantly lower than in other land use systems. This variation might be attributed to the amount of soil organic matter in the forest and grazing land, the depletion of SOM, and the leaching of N in the form of nitrate ions (NO_3^-) from cultivated fields. The amount of SOM present in the soil is a key factor for the presence of TN in adequate amounts, as evidenced by a significant and positive correlation of TN with SOM ($r=0.908$, $p<0.01$). Similar findings were reported by Feleke et al. (2019), who demonstrated that the content of TN in protected grassland was higher than in unprotected grassland because of the variation in the contents of SOM in these land use types.

In this study, in all land use systems and locations, the mean values of C:N ratios were less than 20:1, indicating the mineralization process. It also demonstrated the potential impacts of different land use systems and elevation gradients on the C:N ratio. The lowest value was found in the cultivated land in the upper part of the watershed. This could be attributed to further decomposition and removal of SOM as a result of intensive cultivation and soil erosion. On the other hand, the highest

mean value was found in the forest land of the same elevation due to its high SOC content. This result is in agreement with Seifu et al. (2020), who explained that the highest C:N ratio is in the grazing land where there is high SOM and the lowest value is in the bare land.

The content of available phosphorus could be influenced by land use systems, soil management strategy (application of external inputs), and the amount of soil organic matter. The highest mean value of available phosphorus was observed in the agricultural land as compared to other land use systems in the study area. This might be attributed to long-term applications of phosphorus-containing fertilizer (DAP), which was confirmed during our field survey work and data gathered from farmers and experts. On the other hand, the lowest value in the grazing land might be due to the low turnover of grass biomass to the soil because of overgrazing. This in turn results in lower soil organic matter and organically bound phosphorus, which could be released into the soil solution through mineralization. In forest land, available phosphorus is higher than in grazing land because of better accumulation of SOM. Parallel with this result, Seifu et al. (2020) reported the highest mean value of available phosphorus in the cultivated land as compared to the forest and grazing land use systems in the semiarid watershed in northwestern Ethiopia. Abera and Assen (2019) also reported similar results from the Wanka watershed in northwestern Ethiopia. Likewise, De et al. (2022) and Tellen and Yerima (2018) explained the higher content of available phosphorus and nitrogen in cultivated land because of the continuous application of chemical fertilizers. The high availability of phosphorus is an indication of low p-fixation, which is highly dependent on soil pH (availability decreases in both acidic and alkaline soil conditions). The content of SOM also determines the availability of phosphorus. The findings of other researchers (Stephens et al. 2014; Feleke et al. 2019) indicated that phosphorus is positively and significantly correlated with SOM and that its availability is dependent on the mineralization of SOM. However, in the current study, phosphorus was significantly but negatively correlated with SOM. The reason might be due to the continuous application of phosphorus fertilizer in the cultivated field; conversely, the lowest value of SOM was found in the same field.

Exchangeable bases (Na^+ , K^+ , Ca^{2+} , and Mg^{2+}), CEC, and PBS

The lowest values of exchangeable bases except Mg^{2+} were found in the cultivated land, and this might be due to intensive cultivation, leaching of basic cations, soil erosion, low SOM content, and removal of cations with crop harvest from this land use type. On the other hand, the highest value of CEC was in the forest land, which could

be linked with the amount of SOM. Exchangeable K^+ and exchangeable Ca^{2+} were positively and significantly ($r=0.645$, $p<0.01$ and $r=0.459$, $p<0.01$) correlated with SOM; exchangeable Na^+ and Mg^{2+} also showed a positive correlation with SOM ($r=0.154$ and $r=0.060$). From the results of exchangeable bases, it is observed that the exchange sites of the colloidal particle were dominated by divalent cations (Ca^{2+} , and Mg^{2+}). According to Bohn et al. (2001), for agriculturally productive soils, the order of exchangeable bases should be $\text{Ca}^{2+} > \text{Mg}^{2+} > \text{K}^+ > \text{Na}^+$, and any deviation from this could cause a cation imbalance for plants. Another important point is the ratio of Ca to Mg cations (Ca:Mg). For most crops, the optimal Ca:Mg ratio should be between 3:1 and 4:1 (Landon 2014). If the ratio is less than 3:1, it will inhibit the uptake of phosphorus.

In the present study, the higher content of CEC in forest land followed by grazing land might be attributed to the content of SOM. This could be evidenced by a positive ($r=0.31$, $p>0.05$) correlation of CEC with SOC. In agreement with this result, Abera and Assen (2019) demonstrated that the CEC in the natural forest land was higher than in grazing and cultivated lands because of the high SOM content in the forest land. The content of CE also varied along the top sequence of the watershed. The highest value was found at the lower part; this could be due to the large coverage of Vertisol and Leptosol (soil distribution map), with clay minerals having a high surface area. This could be substantiated by a positive and significant correlation ($r=0.548$, $p<0.01$) of CEC with clay fraction (Table 7). In line with this result, Elias (2019) reported high CEC in Vertisols and Leptosols, which are dominated by smectite minerals with high surface areas and pH-dependent charges, in the cultivated fields of the Ethiopian highlands.

Soil quality status under different land use systems

The soil quality of the Suha watershed is declining because of anthropogenic activities, as evidenced by the negative values of SQDI for most soil attributes under cultivated and grazing lands. In these land use systems, ISS, SOC, TN, and exchangeable K are the most deteriorating soil quality indicators. This might be due to soil erosion, leaching, depletion of SOC as a result of intensive cultivation, low returns of organic inputs, and poor management. On the other hand, high positive values of available phosphorus and clay fraction in the cultivated field could be because of the regular application of chemical fertilizer and cultivation, which enhances the weathering process and hence increases clay fraction. The total SQDI values also showed that soil conditions under cultivated and grazing land use systems have been experiencing progressive deterioration as a result of poor

management strategies. These results agreed with the findings of Abera and Assen (2019), who reported declining soil quality indicators in disturbed ecosystems (agricultural land and grazing land) in the Wanka watershed, northwestern Ethiopia. Feleke et al. (2019) also reported that a higher value of SQI was found in protected grassland and a lower value in unprotected grassland in Farta District, northwestern Ethiopia. Other researchers (Eyayu et al. 2009; Gui et al. 2009) also reported similar findings.

Conclusions

The present study was conducted to evaluate the impacts of land use and cover change on soil quality in terms of its physical and chemical indicators along the top sequence of the Suha watershed in north-western Ethiopia. The results illustrated that land use and land cover change (particularly from forest land to agricultural land) had detrimental impacts on soil physical and chemical quality indicators that resulted in the deterioration of soil conditions and a reduction in agricultural productivity. Soil quality indicators that are sensitive to land use change and management are below their threshold values, indicating that soil condition is deteriorating. Poor management of agricultural lands is the main factor in the decline of soil quality indicators, which could be explained by the depletion of soil organic carbon (SOC) and nitrogen deficiency. In addition, the soil quality degradation index values of most soil quality indicators are highly negative in the cultivated field, indicating the existence of poor management and deterioration of soil conditions. Moreover, aggregated soil quality index values in the agricultural land use systems are below the optimum range, which is evidence of poor management in these land use systems. Therefore, soil and site-specific management strategies are required to reverse degraded soil conditions and maintain soil quality indicators to improve soil ecosystem services, enhance agricultural productivity, and ensure food security. In this regard, the application of integrated soil fertility management (ISFM) strategies and soil conservation technologies are suggested as remedial actions. Land use and land cover change are two of the major factors that impact soil quality; other factors such as soil types, slope classes, and soil management strategies should be explored to design and implement holistic and effective soil management strategies.

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Author contributions

NY: conceptualization, design methodology, data collection and analysis, writing original draft, EE: conceptualization, design methodology, supervision, validation, editing the manuscript. GLF: conceptualization, design methodology, supervision, validation, editing the manuscript. All authors read and approved the final manuscript.

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Availability of data and materials

All the data used in this research are included in the manuscript.

Declarations

Ethics approval and consent to participate

Not applicable.

Consent for publication

All authors agreed and approved the manuscript for publication in *Environmental Systems Research Journal*.

Competing interests

The authors declared that they have no financial or non-financial competing interests.

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