Esa *et al. Environ Syst Res (2018) 7:21*
https://doi.org/10.1186/s40068-018-0122-0

Environmental Systems Research

RESEARCH

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Implications of land use/cover dynamics on soil erosion potential of agricultural watershed, northwestern highlands of Ethiopia

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Abstract

Background: Assessment of soil loss rates is crucial to sustainably enhance the benefits of land resources and diminish the adverse impacts of land degradation thereby areas requiring immediate soil erosion management practices can be identified. The study aimed to examine the impacts of land cover dynamics on the spatiotemporal patterns of erosion hotspots. RUSLE factors were produced using GIS and remote sensing techniques.

Results: The RUSLE model adapted to Ethiopian conditions was run for 2004 and 2014 where input data layers were overlaid. The results of the model showed clear patterns of changes characterized by gradual shifting of one erosion soil loss severity class into next higher class. There was a net increase in the total soil loss largely under the very high, low and very low soil loss severity classes by 8%, 21% and 9% despite a decline in other severity classes, respectively. It also revealed that more than two-third of the catchment has experienced soil losses rates higher than the tolerable value reported for Ethiopia over which agriculturists should be concerned.

Conclusions: Therefore, the observed soil loss rate and sediment yield in the study catchment would lead to further ecological deterioration unless site-specific participatory watershed management practices are employed.

Keywords: Land cover dynamics, RUSLE, Soil loss tolerance, Erosion hotspots, Geo-spatial technologies

Background

Human-induced land cover changes (LCCs) has become a serious environmental problem over centuries (Sharma et al. 2011). In the Ethiopian highlands, favorable agro-climatic conditions have attracted early human settlers to occupy all agriculturally suitable areas over the past several centuries (Hurni 1988). Accordingly, agriculture gradually expanded from gently sloping land into steeper slopes in the highlands as well as the flat swampy plains of the plateau with the subsequent clearing of forests and other vegetation. The removal of vegetation cover and associated expansion of the traditional farming system in the Ethiopian highlands are the major causes of perturbations in the hydrological cycle and triggered soil erosion processes (Mekuria 2005).

Soil erosion has become a threat to sustainable agricultural production and water quality (Prasannakumar et al. 2012). It is one of the physical processes of land degradation widely spread in the highlands of Ethiopia (Hurni 1988). It causes irreversible damage to the natural resource base in the present day of Ethiopia (Hurni 1990). Studies in Ethiopia indicated that 57% and 28% of the area are moderately and severely affected by soil erosion (Lambin and Geist 2006). The highest soil erosion rates has occurred in the western areas where the high amount of rainfall is recorded than in the relatively low rainfall regions of the northern, central and eastern parts of Ethiopia (Hurni 1988). In the highlands of Ethiopia, rates of annual soil loss reached as high as 200–300 t ha^{-1} $year^{-1}$, which could amount to 23.4×109 tones of soil annually (Hurni 1993). However, the severity of soil erosion increases on steeper topographic position and poor vegetation cover (Abate 2011).

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The rates of erosion process are strongly governed by anthropogenic factors such as land use and management practices, land tenure, and institutional support systems (Mekuria 2005). Shrinkage of grassland and the resultant loss of available fodder for livestock have caused overgrazing and consequent conversion of grasslands into degraded lands (Gete and Hurni 2001). Expansions of farmlands at the expense of other land cover classes and traditional farming practices are encouraging erosion and loss of available nutrients. These include cultivation of cereal crops such as teff (*Eragrostis tef*) and wheat (*Triticum aestivum*), which require the preparation of a finely tilled seedbed, the single cropping of fields, and drainage ditch to facilitate drainage in croplands during rainstorms (Badege 2001).

The results of long-term small test plots and micro-catchments analysis of the upper Blue Nile Basin indicated that surface runoff and sediment yield increased due to intensified land use and land degradation induced by population pressure over the past 30–50 years (Hurni et al. 2005). As a result, soil erosion is greatest on cultivated land, where the average annual loss is 42 t ha^{-1} $year^{-1}$ and accounted nearly half of the soil loss as compared with 5 t ha^{-1} $year^{-1}$ from pasture lands (Hurni 1988, 1993). In some cases, the average soil loss from croplands in the highlands of Ethiopia reached as high as 100 t ha^{-1} $year^{-1}$ (Mati et al. 2000) and 130–170 t ha^{-1} $year^{-1}$ (Hurni et al. 2005). However, for the entire Ethiopian highlands erosion rate was estimated to be 35 t ha^{-1} $year^{-1}$ (Mitiku et al. 2006). Research result based on plot level measurement in the Ethiopian highlands also confirmed that soil erosion was ranged from 0 to 170 t ha^{-1} $year^{-1}$ (Gete and Hurni 2001). Soil loss rates based on Soil Conservation Research Project (SCRP) stations in Amhara Region also showed rates between 0.04 and 212 t ha^{-1} $year^{-1}$ as well (Lakew et al. 2000).

Universal Soil Loss Equation (USLE) and its revised form i.e., Revised Universal Soil Loss Equation (RUSLE) are principally used to estimate the rate of soil loss from the landscape and guide the priority areas of conservation practices to a soil loss tolerance (SLT) level. SLT is the average erosion rate that can occur with little or no long-term degradation of the soil with values ranging from 5 to 11 t ha^{-1} $year^{-1}$ (Renard et al. 1996). Studies in northwestern highlands of Ethiopia reported that the mean SLT value was 6–10 t ha^{-1} $year^{-1}$ where soil erosion value below this is assumed not to be a problem of sustainability (Hurni 1983). Morgan (2005) also estimated the average African SLT value at the rate of 10 t ha^{-1} $year^{-1}$ over which agriculturists should be concerned.

The study area has experienced spatiotemporal land cover dynamics. These changes were largely caused by unsustainable land use practices such as overgrazing, expansions of farmlands at the expenses of other land cover classes, and deforestation. The agricultural practices, in the study area, are largely characterized by small-scale, fragmented and traditional tillage with low fertility level. Alternatively, farming operations are usually performed during intense rainfall events where weak soil surface caused by tillage and absence of vegetative cover exposing farmlands to direct rainfall impact and hence, increased stream loads. During higher rainfall months (June to September), the mainstream is a source of flooding along the lower courses of the mainstream channel, and sedimentation into low-lying areas and Lake Tana. Water hyacinth (*Eichhornia crassipes*) is also expanding along the shoreline of the lake. Thus, impacts of land cover dynamics on quality of environmental elements (e.g., soil quality and quantity) are of widespread concern and a great challenge to researchers and policymakers. The result of this research is crucial to sustainably enhance the benefits of land resources and diminish the adverse impacts of land degradation. Therefore, the aim of this study was to examine the spatiotemporal variations of erosion hotspots and mean annual sediment delivery in the catchment and identify areas requiring immediate soil erosion management practices.

Methods

Description of the study area

The Gelda catchment is located between 37°25′55″E and 37°41′30″E longitude and 11°38′15″N to 11°46′15″N longitude. It is almost entirely found in Amhara region, South Gondar administrative zone, Dera wereda. The study area has 26,264 ha of land, covering about 2.2% of the Lake Tana watershed. The study catchment is drained by Gelda River that flows into Lake Tana (Fig. 1). The landform of the catchment reflects its geological history where volcanic uplifting has created initial landmass elevation, surface fracturing and subsequent outpouring of basaltic lava that provided a thick protective cap (Eleni et al. 2013). According to the report of Geological Survey of Ethiopia (GSE 1996), the catchment generally comprises materials ranging from alkaline to transitional basalts that often forming shield volcanoes, with minor trachyte and phenolite flows called "Tarmaber Guassa Formation" in the southeast and eastern parts of the catchment. These geological structures were formed during the Oligocene to Miocene epochs of the Tertiary period. The report also revealed that the western and northwestern parts of the catchment consist of alluvial and lacustrine deposits of the Quaternary period.

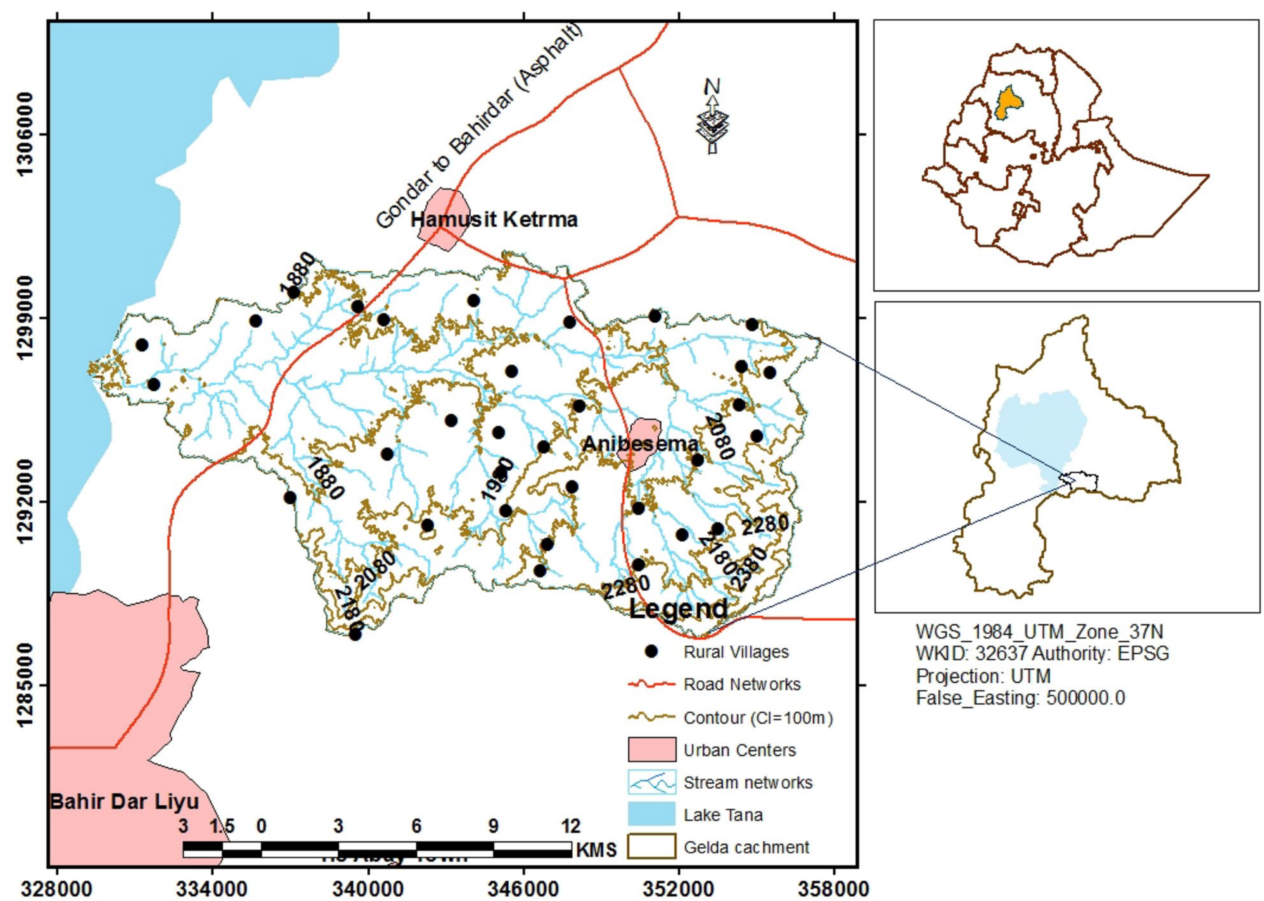

Fig. 1 Location map of Gelda catchment, Northwestern highlands of Ethiopia

The altitude ranges from 1780 to 2481 m above sea level. According to FAO (2006), the gradient is gently sloping (0–7%) comprising 48.5% (12,682 ha) and strongly sloping (8–16%) with 37.13% (9714 ha) areal coverage of the catchment. The moderately steep (16–30%) and steep slope (> 30%) that cover about 11.22% (2936 ha) and 3% (787 ha) respectively, are commonly found in the south-east and southern corners of the catchment. Slopes more than 16% are largely found along the south-east and southern margins of the catchment, locally called Senemariam, Debresina, and Laguna upland areas. Gleysols (54.9%) and Nitisols (30.5%) constitute the major soil types of the Gelda catchment. However, Gleysols are poorly drained with seasonal water accumulation (FAO 1997). The other soils types such as Regosols and Luvisols are commonly found on the sloping lands.

A reliable long-term meteorological data was available in Bahirdar meteorological station, temperature and rainfall records were considered in determining micro-climate of the study catchment. Generally, the climate of the area is generally subtropical with average total annual rainfall of 1453 mm (Hurni 1998). The long-term rainfall values for the period between 1961 and 2014 of the study area indicated the mean monthly standard deviation of 46.53 mm indicating the high seasonal variability of monthly rainfall records over the time span of 54 years (Fig. 2). The long-term mean total annual rainfall records and standard deviation (217.6 mm) also reflected a general declining trend of total rainfall records 26 mm per annum and high inter-annual rainfall variability over the time span of 54 years (Fig. 3). The available mean monthly rainfall records also showed a predominantly unimodal pattern. According to Daniel (1977), the rainfall coefficient value for mean monthly rainfall records indicated that June (1.5), July (3.6), August (3.3), September (1.7), and October (0.7) are the main rainy season, which accounted about 90% of the mean annual rainfall (Fig. 2). The highest mean monthly temperature (30 °C) is recorded in April whereas the minimum (8 °C) in December with a mean monthly range of 5 °C (Fig. 2).

Generally, the study catchment comprises moist Weynadega (largest coverage) and moist dega agroecology

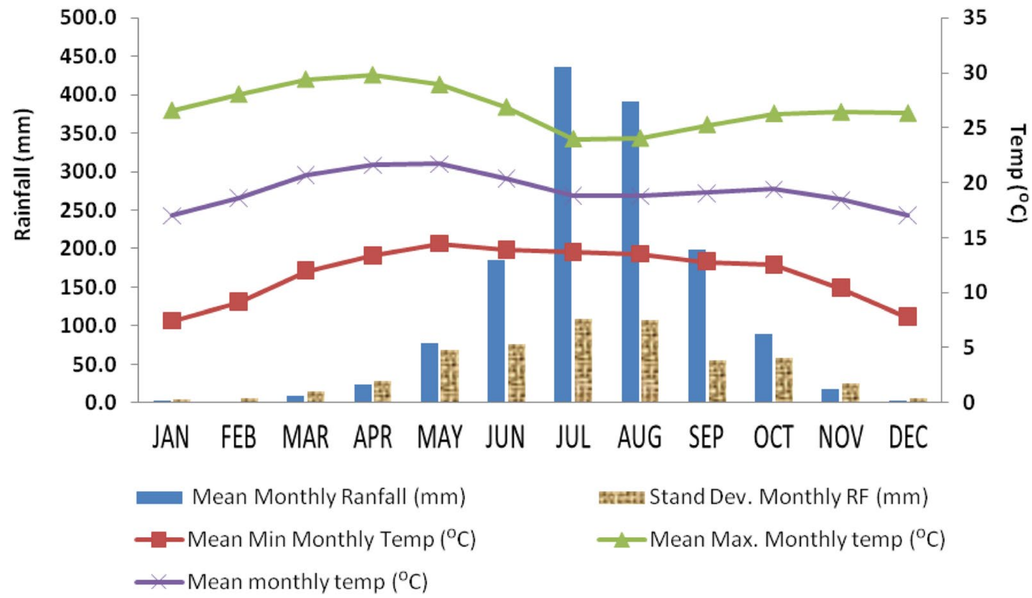

Fig. 2 Mean monthly rainfall and mean monthly temperature records of study area. It is based on the records of nearby *Bahirdar* station (1961–2014) from National Meteorological service Agency, Ethiopia

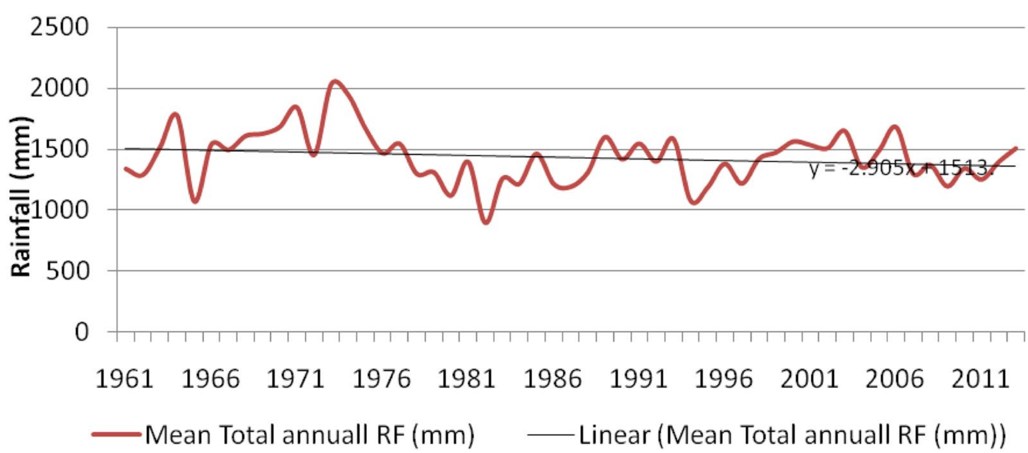

Fig. 3 Long-term mean total annual rainfall records of the study area. It is based on the records of nearby *Bahirdar* station (1961–2014) from National Meteorological service Agency, Ethiopia

zones characterized by average length of growing period ranging from 120 to 240 days (Hurni 1998). According to Hurni (1998), the local agroecology is dominated by *weynadega*. It is ecologically most suitable for rainfed farming, and to the cultivation of various crops such as teff (*Eragrotis teff*), nuog (*Guizotia abyssinica*), maize (*Zea mais*) and others. According to Dera Woreda ARD office (2013), vegetation species largely dominated by *Juniperus procera* (Habesha tid) are commonly found around churches and grave yards. Others such as *Hagenia abyssinica* (kosso); *Albizia amara* (sassa), *Podocarpus falcatus* (zigba); *Cordia africana* (wanza); and *Ficus vasta* (warka) are also found in the study catchment. However, as confirmed with field observation, the non-indigenous tree species like *cypress* (yeferenj tid), *Acacia sieberiana* (Yeferenj girar) and *Eucalyptus* spp. (Bahir zaf) are expanding.

Methodology

Soil loss estimation

Soil loss estimation is affected by complex factors of erosion and the existing availability of data at regional scale

(Sharma et al. 2011). Despite the difficulty of precise estimation or prediction of erosion caused by the complexity of the variables, advances in geospatial technology have presented efficient monitoring, analysis and management methods of land resources (Woldeamlak 2003; Woldeamlak and Sterk 2005; Prasannakumar et al. 2012; Berhan et al. 2015). Quantitative assessment of soil loss is required to infer the extent and severity of erosion problems so that efficient land management strategies can be developed and implemented.

In recent times, various studies have been carried out to estimate the potential implications of LULC dynamics on soil erosion hazard at different spatiotemporal scales (Brath et al. 2002; Sharma et al. 2011; Gizachew 2014). In Ethiopia and many other developing countries where basic data for erosion assessment is not adequate, robust soil loss models with the available resources are employed for the analysis (Kaltenrieder 2007). Accordingly, a number of parametric models like USLE and its revised forms are the most widely used empirical equations to estimate annual soil loss from agricultural lands (Wischmeier and Smith 1978; Renard et al. 1996; Prasannakumar et al. 2012). USLE is one of the many models to examine the impacts of land cover on soil erosion potential.

The USLE was developed by Wischmeier and Smith (1978) and later revised into RUSLE by the incorporation of the influence of profile concavity/convexity and improved empirical equation for LS-factors (McCool et al. 1995; Jones et al. 1996; Renard et al. 1997; Kaltenrieder 2007; Renard et al. 2011). Although it is an empirical model, it can predict soil loss rates of ungauged watersheds using information of watershed characteristics and local hydroclimatic data with practical costs and better precision over larger areas (Prasannakumar et al. 2012). Therefore, RUSLE was run between 2004 and 2014 to examine the implications of land cover dynamics on soil loss rates by making all parameters in the equation remained the same except for cover factor for the reference years. The RUSLE model based estimation of soil loss was carried out according to the following methodological framework (Fig. 4).

The RUSLE equation is a function of five input factors in raster data format, such as rainfall erosivity; soil erodibility; slope length and steepness; cover management; and support practice. These factors vary over space and time, and depend on other input variables. Therefore, the RUSLE model used to analyze the soil loss of the catchment is expressed as (Wischmeier and Smith 1978):

$$A = R \times K \times LS \times C \times P \quad (1)$$

where, A is amount of soil loss calculated in tons per hectare per year, R is rainfall factor (in mega joules millimeter per hectare per hour per year), K is soil erodibility factor (ton hectare hour hectare^{-1} megajoule^{-1} millimeter^{-1}),

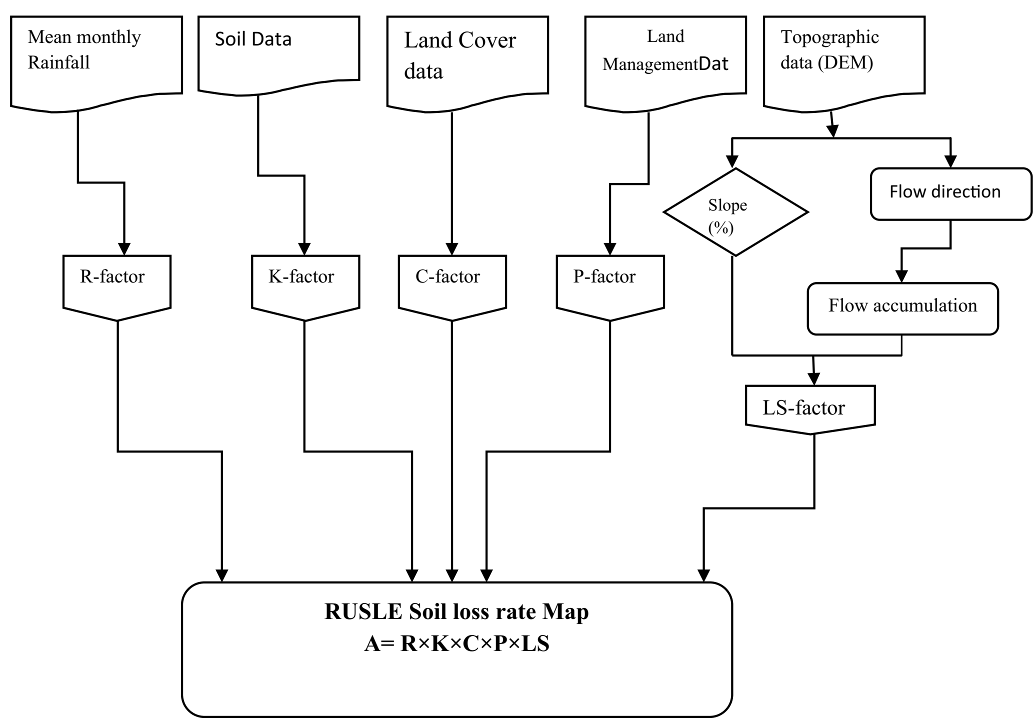

Fig. 4 Flow chart showing the factors of RUSLE model to analyze the potential soil loss of the study catchment with the application of GIS tools

L is slope length factor, *S* is slope steepness factor, *C* is cover factor, and *P* is erosion control practice factor.

The RUSLE model was run for 2004 and 2014 separately. The five raster layers were produced from the attribute values of the RUSLE model and processed by overlay analysis to generate the annual soil loss rate of each cell using "raster calculator" of Spatial Analyst Tool. During each model run, all parameters remained the same except values of the C-factor, which was changed according to the land cover of the respective year.

Rainfall erosivity factor (R-factor)

Rainfall erosivity is the energy of a given storm that depends on rainfall intensity and amount of precipitation, i.e., the value is estimated using the EI_{30} measurement (Renard et al. 1996). Soil erosion is closely related to rainfall partly through the detaching power of raindrops, which strikes the soil aggregates and the contribution of rain to runoff. However, rainfall intensity data are not available in Ethiopia (Hurni 1985). As a result, the empirical equation developed by Hurni (1985) that estimates R-value for the Ethiopian highlands has been used for this research. The R-value is estimated using the following equation:

$$R = -8.12 + 0.562P \quad (2)$$

where R is the rainfall erosivity factor in MJ mm ha^{-1} h^{-1} $year^{-1}$ and P is the mean annual rainfall in millimeters (Hurni 1985).

Annual erosion rates are heavily dominated by single rainfall periods (one or a couple of storms during the rainy season) in the study area. For this study, mean annual rainfall data of four stations found in and around the study area, between 2004 and 2014, were obtained from National Meteorological Service Agency and used for the analyses. As a result, mean annual rainfall of the four stations were used to describe the spatiotemporal soil erosion patterns for the 10 years study period where the land cover map is used to determine C factors. This would be important to get an adequate coverage of different rainfall intensity of the catchment and increase reliability of rainfall values. This means the amount and intensity of rainfall is variable across the watershed.

The spatial interpolation techniques were used along with rainfall data for meteorological stations based on ground control points (GCPs) of meteorological stations for assessing the spatial variability of rainfall and rainfall erosivity. In order to make the R-factor value most reliable, the spatial distribution of R was calculated from the available rainfall data by considering that the area experiences relatively uniform rainfall within using ordinary kriging interpolation method. This method effectively transforms station data into surface data. The individual average R-values interpolated was used for further RUSLE calculation. Finally, the mean rainfall erosivity factors (R) of each meteorological stations were found to be in the range of 572.05–978.09 MJ mm ha^{-1} h^{-1} $year^{-1}$ while mean R-factor was observed to be 808.38 MJ mm ha^{-1} h^{-1} $year^{-1}$ for the entire study area (Table 1). Therefore, Fig. 5 represented the isoerodent map derived from average R-values of stations using ordinary kriging interpolation technique in *Spatial Analyst Tool* of ArcGIS.

Table 1 Mean annual R-factor values for four stations found within and nearby stations (MJ mm ha^{-1} h^{-1} $year^{-1}$)

	Arbgebya	Anbesame	Hamusit	Bahirdar
Mean annual rainfall	1032.3	1670.9	1754.8	1353.4
R-factors	572.05	930.92	978.09	752.47

Soil erodibility factor (K-factor)

The K-factor is a measure of the natural susceptibility of a given soil to erosion under normal condition of the USLE unit plot maintained in continuous fallow (McCool et al. 1995; Prasannakumar et al. 2012). Different soil types have variable susceptibility to erosion. Fine textured clay soils and coarse textured sandy soils have low K values; medium textured silt and loam soils have moderate K values, while soils with high silt content have high K values due to their inherent physical property of soils (McCool et al. 1995; Renard et al. 1996). Similarly, soil organic matter reduces K values because it produces compounds that bind soil particles together and increases aggregation by reducing detaching effect of raindrop, increasing infiltration and reducing surface runoff (Wischmeier and Smith 1978; Renard et al. 1996, 2011). However, the K-factor determined by soil color in Hurni (1985) is often not very precise given the fact that soils of the same color have quite different K values (Kaltenrieder 2007). Therefore, K is a function of particle size, drainage potential, structural stability, organic matter content, and cohesiveness.

In this study, K-factor is reflecting soil properties concerning their effect on the erodibility of the particular soil as expressed in t ha^{-1} J^{-1} mh^{-1}. For the analysis of K factor, standard digital database for soil type classification was obtained from Ministry of Agriculture and Rural Development of Ethiopia (FAO 1997). According to FAO soil classification, Dystric Gleysols, Dystric Nitisols, Eutric Nitisols, Orthic Luvisols and Eutric Regosols were identified in the study area. The different soil texture and organic matter values that were obtained from laboratory analysis of the study area for the same soil type were averaged and used to determine the K-factor of each soil type as adapted from Stone and Hilborn (2012).

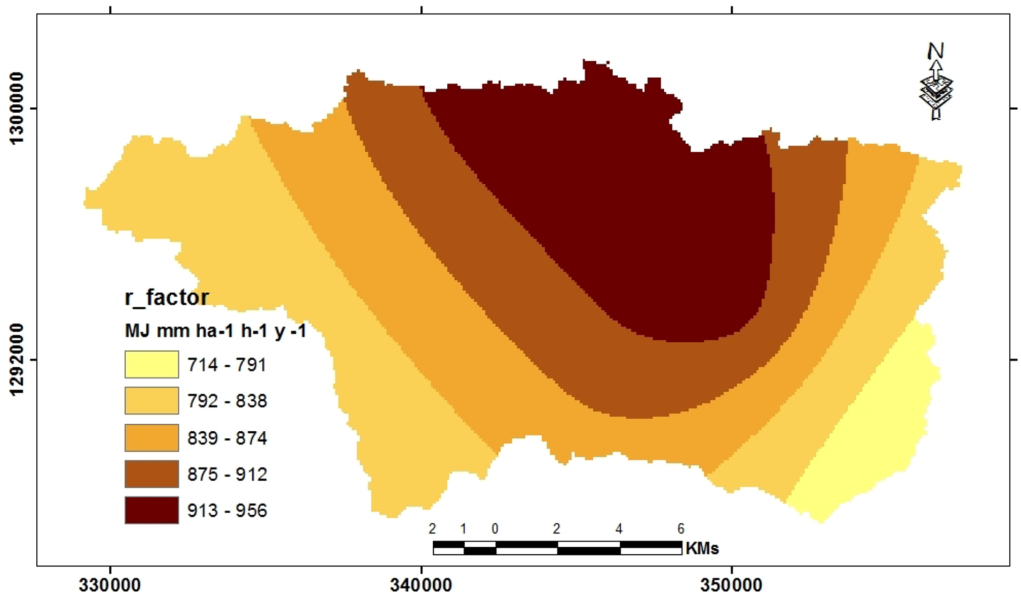

Fig. 5 Isoerodent map of Gelda catchment, Northwestern highlands of Ethiopia

Finally, the vector format soil map was changed into grid and the grid dataset was reclassified with a cell size of 30 m × 30 m resolution into the corresponding K values using Spatial Analyst Tool of ArcGIS 10 (Table 2). The results of analysis indicated that K-factor values of the study area were ranged from 0.20 to 0.30 with the mean value of 0.21 (Fig. 6).

Land use/land cover management factor (C-factor)

The crop management factor represents the ratio of soil loss under a given crop to that of the base soil reflecting the effect of cropping and management practices on the soil erosion rate (Wischmeier and Smith 1978; Renard et al. 1996). It determines the relative effectiveness of soil and crop management systems in preventing soil loss. Studies in highlands of Ethiopia and Eritrea indicated that the density of the crop cover is of crucial importance to determine the rainfall erosivity (Kaltenrieder 2007). Under low (0–30%) and moderate (30–60%) plant cover, all storms can cause erosion during high rainfall erosivity periods than high (> 60%) plant cover. As a result, the C-values can vary from near zero for a very well-protected soil to 1.0 in barren soils before plant growth and 1.5 for a finely tilled surface that produces much runoff and leaves the soil and highly susceptible to rill erosion (Kim and Julien 2006; Benzer 2010; McCool et al. 1995).

Currently, due to the variety of land cover patterns with spatiotemporal variations, remote sensing data sets were used for the assessment of C-factor (Prasannakumar et al. 2012). Since the correlation between measured and predicted soil loss was 0.9 (very high), C values for different land cover types adapted by Hurni (1985) is an excellent tool to determine the C-factors. As a result, the land cover maps of the study area, prepared from Landsat ETM and OLI images for the year 2004 and 2014 (Table 3), were employed for determining the C-factors

Table 2 K-factor determined from soil organic matter and texture analysis data of Gelda catchment using Stone and Hilborn (2012) technique

No	Soil type	Particle size (%)			Textural classes	SOM (%)	K-factor
		Sand	Clay	Silt			
1	Dystric Gleysol	34.6	41.7	23.8	Clay	2.6	0.21
2	Dystric Nitisol	28.3	35.7	36.0	Clay loam	2.8	0.28
3	Eutric Nitisol	48.3	29.7	22.0	Silty clay loam	3.5	0.30
4	Orthic Luvisol	52.0	16.0	32.0	Loam	2.8	0.20
5	Eutric Regosol	33.0	25.0	42.0	Loam	2.6	0.20

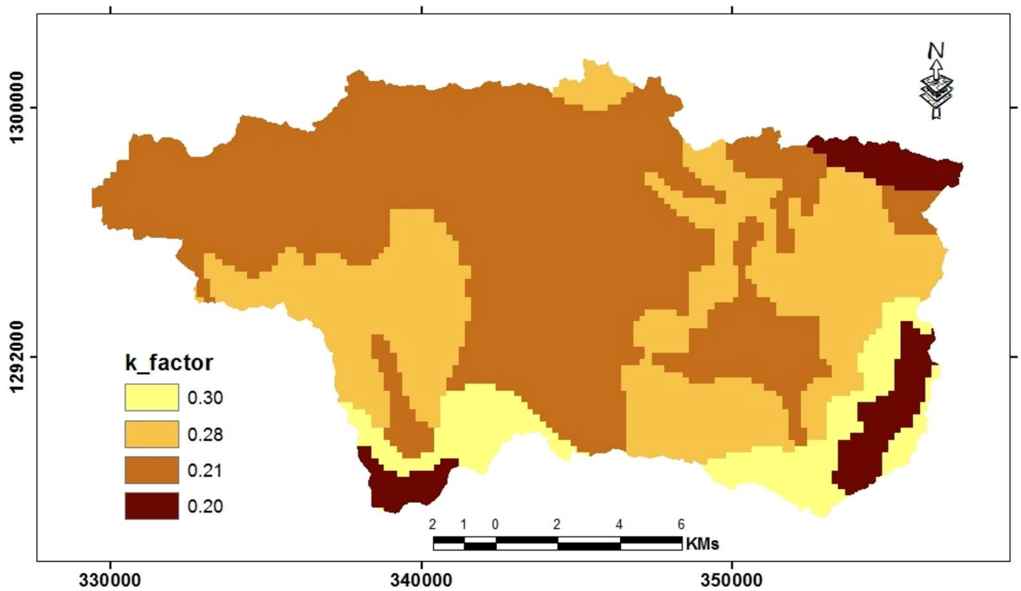

Fig. 6 Soil erodibility (K-factor) map of Gelda catchment, Northwestern highlands of Ethiopia

Table 3 Land use and cover patterns of Gelda catchment between 2004 and 2014, Northwestern highlands of Ethiopia

No	Land cover classes	2004		2014	
		Area (ha)	%	Area (ha)	%
1	Farmland and settlement	13,882.86	52.86	14,260.95	54.3
2	Grassland	7933.32	30.2	1931.67	7.35
3	Shrubs	2915.28	11.1	8464.5	32.23
4	Forest cover	1247.4	4.75	959.31	3.65
5	Bare soil cover	286.11	1.09	648.54	2.47
	Total area	26,264.97	100	26,264.97	100

(Table 4). These were, latter, reclassified into C-factor maps to examine changes in soil loss rates over the studied period (Fig. 7). The classification statistics in ArcGIS revealed the mean C-factor values of 0.24 for 2004 and 0.28 for 2014.

Erosion support practice factor (P-factor)

The need for soil conservation is estimated from the protection factor (P) required for reducing soil erosion from its average rate on unprotected land to the tolerable estimated rate (Kassam et al. 1992). The P-factor is the soil loss ratio with a specific support practice to the corresponding soil loss with up and down slope tillage (Renard et al. 1996). It reflects the impact of supporting practices on the erosion rate. The most commonly used supporting practices are contour tillage, strip-cropping on the contour and terrace systems in which all requires stabilized waterways for the disposal of excess rainfall (Stone and Hilborn 2012). These management activities depend on the slope of the area (Wischmeier and Smith 1978). Although vast area of the catchment has been

Table 4 The land cover classes and the corresponding C-factors of 2004 and 2014 at *Gelda* catchment, NW highlands of Ethiopia. Modified from Hurni (1985) and Prasannakumar et al. (2012)

No	Land cover classes	C-factor	2004		2014	
			Hectares	%	Hectares	%
1	Farmlands	0.34	13,882.86	52.86	14,260.95	54.3
2	Grasslands	0.03	7933.32	30.2	1931.67	7.35
3	Shrubs	0.23	2915.28	11.1	8464.5	32.23
4	Forests	0.03	1247.4	4.75	959.31	3.65
5	Bare soil cover	1.0	286.11	1.09	648.54	2.47

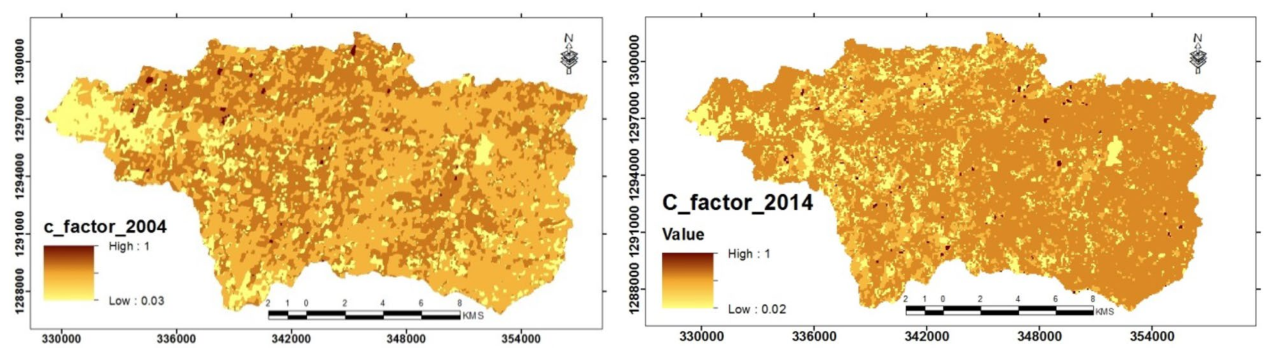

Fig. 7 Land use/land cover management factor maps of 2004 (left) and 2014 (right) at Gelda catchment, Northwestern highlands of Ethiopia

covered with terracing as a response of the recent soil and water conservation programs, only small areas are well treated with appropriate measures while many of them are unsustainable. However, there are varieties of the traditional conservation schemes widely used in the study area. Some of these are contour tillage, compacting seedbeds immediately after tillage for some varieties of crops and the drainage ditches. Drainage ditches and other stabilized waterways on farm plots are often used to safely drain excess runoff from croplands during rainstorms (Bewket and Teferi 2009).

The P-factor values ranges from 0 to 1 depending on the soil management activities employed in an area. According to Prasannakumar et al. (2012), the highest value is assigned to areas with no conservation practices while minimum values correspond to built-up land and plantation area with strip and contour cropping. As a result, the lower P value, the more effective the conservation practices. Evaluation of the practices in RUSLE requires to estimate surface roughness and runoff reduction, but some of the P-factor values are slope dependent (McCool et al. 1995). In this study, there was only a small area that has been left untreated with conservation structures through the agricultural extension campaign of the government. However, the conservation structures are poorly implemented due to their top-down approach. Considering the lack of data in the area on permanent management factors, the P values, suggested by Hurni (1985), Wischmeier and Smith (1978) and Sharma et al. (2011) that considers only two types of land uses (agricultural and nonagricultural) and land slopes, were used in this study (Table 5). In this study, the classification statistics in ArcGIS calculated the mean P value of 0.5.

Table 5 P-values modified from Wischmeier and Smith (1978), Hurni (1985) and Sharma et al. (2011)

Land use type	Slope (%)	P-factor
Agricultural lands	0–5	0.11
	5–10	0.12
	10–20	0.14
	20–30	0.22
	30–50	0.31
	50–100	0.43
Non agricultural land uses		1.00

Slope length and steepness (LS) factor

As one component of topography, local slope gradient (S-sub factor) influences flow velocity and thus the rate of erosion (Kaltenrieder 2007). Slope length (L-sub factor) describes the distance between the origin and termination of inter-rill processes (Wischmeier and Smith 1978; Renard et al. 1996). In RUSLE, the LS-factor represents a ratio of soil loss under given conditions to that at a site with the “standard” slope steepness of 9% and slope length of 22 m plot (Renard et al. 1996; Kaltenrieder 2007). The steeper and longer the slope, the higher is the momentum to generate soil erosion. In this study, the technique for estimating the RUSLE LS-factor is computed based on flow accumulation and slope steepness in degree as proposed by Moore and Burch (1986a, b)

$$LS = ([\text{flow accumulation}] * [\text{cell size}]/22.13)^{0.4} * \left(\sin[\text{slope gradient}]/0.0896\right)^{1.3} \quad (3)$$

where flow accumulation denotes the accumulated upslope contributing area for a given cell, LS = combined slope length and slope steepness factor, with a resolution of 30 m × 30 m grid cell size and sine slope value of slope degree.

In this study, the technique for estimating the RUSLE LS-factor was computed based on flow accumulation and slope steepness in degree as proposed by Benzer (2010):

LS = ([flow accumulation] * [cell size]/22.13)$^{0.4}$ * (sin [slope gradient]/0.0896)$^{1.3}$, where flow accumulation denotes the accumulated upslope contributing area for a given cell, LS = combined slope length and slope steepness factor, with a resolution of 30 m × 30 m grid cell size and sin slope value of slope degree.

This equation was processed with the "raster calculator" of Spatial Analyst Tool in ArcGIS 10 to provide LS-factor value. SRTM image with 30 m × 30 m pixel size resolution was pre-processed and appropriate size of the study area was extracted by Spatial Analyst Tool "extract by mask" technique. The corresponding flow accumulation and slope steepness map was generated in raster format (Fig. 8a and b). Finally, the LS-factor grid was estimated with the equation using spatial analyst "raster calculator". Accordingly, the topographic factor values in the study area that varies from 0 to 19; with mean and standard deviation of about 2.3 and 4.8 were determined, respectively (Fig. 8c).

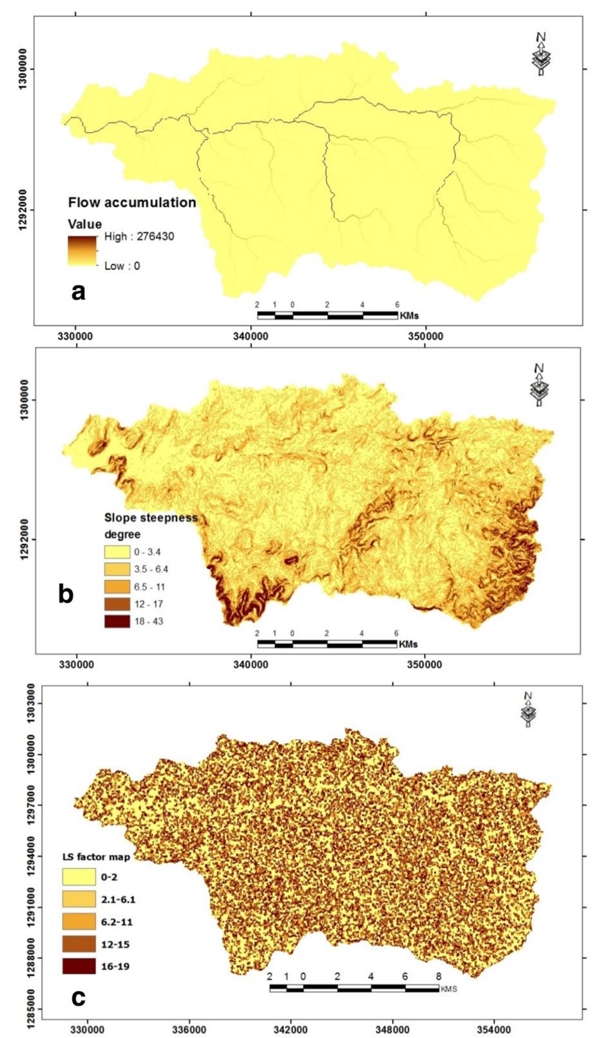

Fig. 8 Flow accumulation, slope steepness, and topographic (LS) factor maps **a**, **b** and **c** (top-down) at Gelda catchment, Northwestern highlands of Ethiopia, respectively

Results and discussion

Estimated soil loss rate

The magnitude of change in the soil erosion potential and their spatial distribution as depicted in Fig. 8. The "*classification statistics*" of ArcGIS revealed that the potential soil loss for the study area was ranging from 1 in low soil erosion-prone areas to about 109 and 128 t ha^{-1} $year^{-1}$ on the steeper slopes and high erosion vulnerable areas between 2004 and 2014, respectively. The estimated average annual soil loss was increased by 16% in this study period, i.e., from 49 to 57 t ha^{-1} $year^{-1}$ between 2004 and 2014, respectively. The increased mean erosion potential for the whole catchment in the studied period reflected the serious effects of different land use transitions. The result of the present research is generally consistent with the findings of previous studies in the highlands of Ethiopia (Hurni 1993; Lakew et al. 2000; Gete and Hurni 2001; Woldeamlak and Sterk 2005; Mitiku et al. 2006; Berhan et al. 2015; Gizachew 2014; Tadesse and Abebe 2014).

Shrinkage of forest cover and grasslands were observed, while farmlands and bare soil cover classes showed expansions from their original extent between 2004 and 2014 (Table 3). The expansion in bare soil cover was largely attributed to area gained from grasslands and farmlands. This could possibly be the impacts of poor farming practices and overgrazing of grasslands. Conversely, conversions of the original forest cover into farmlands and shrubs caused a decline in forest cover. Similarly, reduction in grassland covers was largely caused by the conversion of its initial extent into farmlands. The existence of these spatiotemporal transformations of the existing land cover classes has confirmed strong associations with soil loss rates in this period of analysis. The main reason for this increase in the soil erosion potential of the catchment over the studied period could be attributed to expansions of farmlands and bare soil cover at the expenses of forest area and grasslands. From these results, it can be inferred that any land use transitions into farmlands would be detrimental, as it was the major source of sediment while forest was the most effective barrier to soil loss.

Different researchers use various soil loss severity classes worldwide depending on the purpose of the study and the geographical context of the study area. However, the present study employed FAO/UNDP (1984) soil removal and erosion risk prioritization mapping scale applied to the Ethiopian highlands to effectively visualize the spatial distributions of erosion hotspots in the study catchment (Fig. 9). Thus, the raster layers were collapsed into five soil erosion severity classes, namely very high (> 50 t ha^{-1} $year^{-1}$), high (30–50 t ha^{-1} $year^{-1}$),

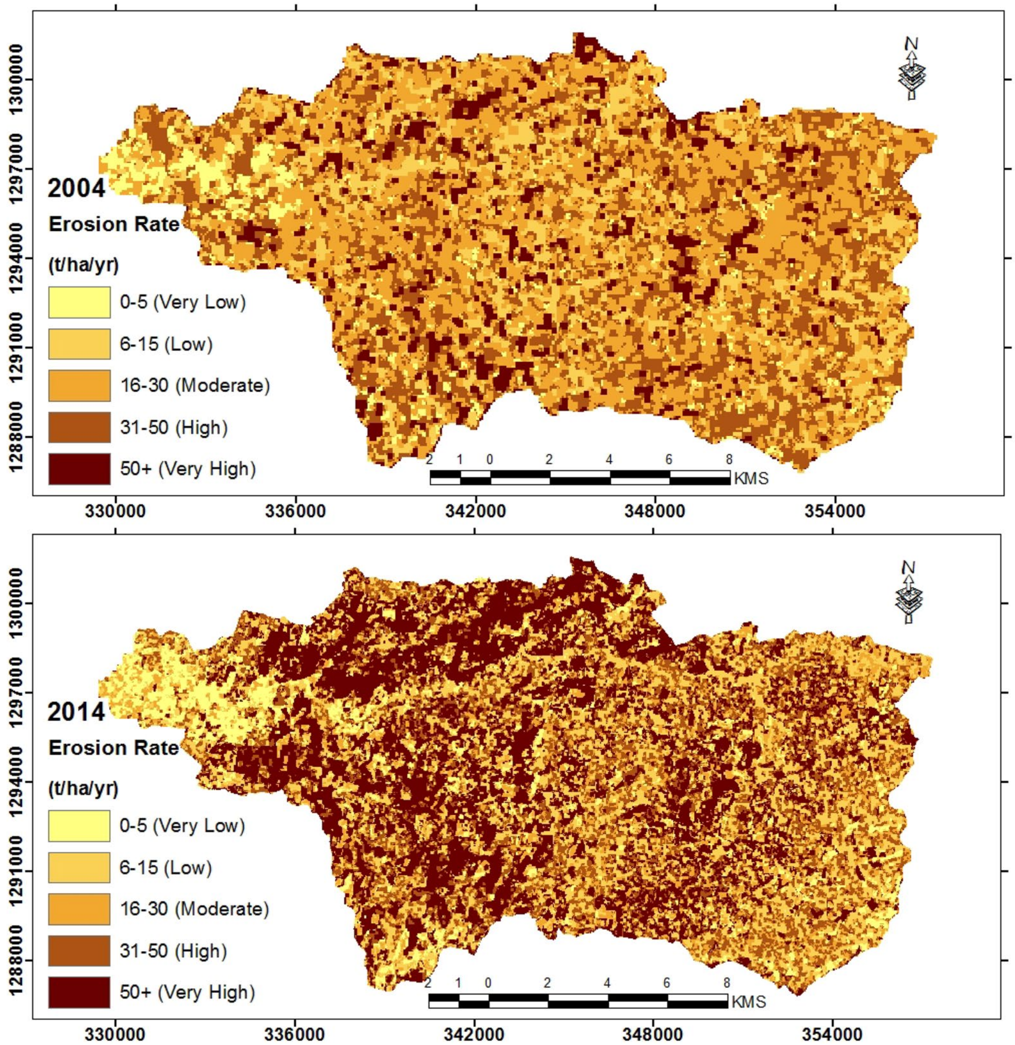

Fig. 9 Spatial distributions of potential annual soil loss and severity classes at Gelda catchment between 2004 and 2014 (top-down), Northwestern highlands of Ethiopia

moderate (15–30 t ha^{-1} year^{-1}), low (5–15 t ha^{-1} year^{-1}) and very low (0–5 t ha^{-1} year^{-1}) for two reference years. The degree of rill and gully formations and development, tree root exposure and severe damage from erosion and sedimentation are some of the indicators used to prioritize soil erosion severity classes (Morgan 2005; Berhan et al. 2015). These could determine the degree of severity of erosion in an area mainly caused by the major factors of soil erosion.

The area coverage and percentage proportion were tabulated for each of the potential soil erosion categories (Table 6). The largest coverage was classified as very high (> 51 t ha^{-1} year^{-1}) followed by high (30–50 t ha^{-1} year^{-1}), moderate (15–30 t ha^{-1} year^{-1}) and low (5–15 t ha^{-1} year^{-1}) soil loss severity classes, which accounted about 32%, 25%, 18% and 19% of the entire study catchment in 2004, respectively. In 2014, about 35%, 22% and 23% of the study area were classified as very high (> 50 t ha^{-1} year^{-1}), high (30–50 t ha^{-1} year^{-1}) and low (5–15 t ha^{-1} year^{-1}) soil loss severity classes, respectively. Table 6 also showed clear patterns of changes characterized by gradual shifting of one erosion severity class into next higher class. As a result, a slight decline in areal coverage of soil severity classes was observed in moderate and high soil loss severity class by 22% and 12%, respectively. However, there was a net increase in the total soil loss under very high, low, and very low soil loss severity classes by 8%, 21% and 9%, respectively.

Table 6 Annual soil loss rate, severity classes and conservation priorities of Gelda catchment, Northwestern highlands of Ethiopia

Rate of soil loss (t ha^{-1} year^{-1})	2004		2014		Between 2004 and 2014		Severity classes	Conservation priorities
	Ha	%	Ha	%	Ha	%		
0–5	1566	6	1706	6	140	9	Very low	V
6–15	4889	19	5933	23	1044	21	Low	VI
16–30	4814	18	3760	14	− 1054	− 22	Moderate	III
31–50	6547	25	5781	22	− 766	− 12	High	II
50+	8448	32	9084	35	636	8	Very high	I

Priority of management requirements

Primarily RUSLE was used for conservation planning by comparing the computed soil loss with SLT value of the study catchment. In the present study, the soil loss rate was compared with the SLT values, i.e., 6–10 t ha^{-1} year^{-1} as it was estimated to the northwestern highlands of Ethiopia by SCRP (Hurni 1983). As a result, the results of the soil erosion analysis showed that about 74.5% and 82.5% of the study catchment has experienced soil loss rate higher than the average SLT level adapted to the northwestern highlands of Ethiopia between 2004 and 2014. However, low rates of soil erosion (< 10 t ha^{-1} year^{-1}) can be compensated to some extent by the formation of new topsoil (> 1.5 mm year^{-1}) in humid and warm environments which is equivalent to an annual addition of more than 12 t ha^{-1} (Kassam et al. 1992). Such data are valuable to provide specific guidelines for carrying out soil conservation within the specified limits over which agriculturists should be concerned (Hurni 1985).

The erosion potential had undergone a great degree of spatiotemporal changes where areas with steep slopes, high drainage density, and high erosion-prone soil experienced high to very severe soil loss classes. These areas were largely located in the mountainous and sub-mountainous regions of the eastern, southern, southwestern margins, and some patches of the catchment with high topographic ruggedness. Poor vegetation cover increased agricultural practices in the high erosion-prone soil, overgrazing and deforestation also aggravated the problem. This is because farming operations are usually performed during intense rainfall events where there is weak soil surface caused by tillage and absence of vegetative cover. Therefore, third to first order soil conservation options would be devised to protect the soil from raindrop impact by improving soil infiltration capacity and moisture content. Increasing surface roughness and decreasing slope length are also essential to reduce runoff momentum and maintain the fertility level of soils. More specifically, both agronomic such as area closure and afforestation, strip-cropping, agroforestry; and mechanical soil conservation measures such as check dams, construction of hillside ditches for stormwater diversion, terracing, and contour tillage would be potential land management options in the area to reduce the power of runoff and soil erosion.

Areas reportedly experienced moderate to low soil erosion classes were dispersed largely on the central and south-western and north-western margins, and some patches at the western end of the catchment over the studied period. These areas require conservation priorities of fourth and fifth order, respectively. In areas under moderate soil erosion severity class, gully reclamation measures, strip cultivation, grass strips and waterway, contour tillage, construction of stone bunds and terraces and control of hillside grazing would be recommended to reduce soil erosion and environmental damage. In low soil erosion severity areas, the rate of soil erosion is not a serious problem of sustainability, however, farming practices that minimize soil erosion could be practiced to enhance soil fertility, maximize the productivity and long-term sustainability the resource use. Therefore, soil erosion risk mapping is generally useful for identifying soil loss hotspot areas and prioritizing land management interventions (Berhan et al. 2015).

Conclusions

One of the major factors for the spatiotemporal variability of soil loss is attributed to LCCs following the expansion of agricultural practices in high erosion-prone soil, overgrazing and deforestation. This is particularly true in the highlands of Ethiopia, where there is a shortage of farmlands mainly driven by population pressure. The larger portion of study catchment has experienced from very severe to high soil erosion and sedimentation problem. The undulating nature of topography and high steep slopes in some areas partly contributed to severe erosion problem and maximum sediment yield generation from the catchment. The mean annual soil loss rate has increased by 16.3 t ha^{-1} year^{-1}, i.e., a net increase of annual soil loss at the rate of about 2.1 × 105 t year^{-1} between 2004 and

2014. It was largely due to the impacts of LCCs on soil erosion in the study catchment. In areas where soil loss rates exceeded the SLT, soil erosion threat is the major environmental concern for sustainable agricultural productivity. The amount of mean sediment transported at the outlet of Gelda River increased by 16% over the entire studied period. This can cause sedimentation and shoreline damage. As a result, land cover dynamics in this period of analysis have largely influenced sedimentation rates as well as the lake's ecosystem negatively.

Despite the impressive achievements made by the soil conservation campaigns in Amhara region since 2011; there is a fear of declined enthusiasm due to top-down nature of the program. These facts call for a concerted action plan for integrated watershed management such as soil and water conservation technologies, area closure, afforestation; agricultural and livestock development that includes controlled grazing and improved agricultural inputs. However, the results of the RUSLE model integrated with geospatial technologies do not provide adequate information to decide about appropriate soil and water conservation interventions in tackling the challenges of soil erosion in the study catchment. Therefore, long-term studies on small test plots and micro-catchment level conservation projects with detailed biophysical and socioeconomic data should be undertaken to guide site-specific land management practices.

Authors' contributions

EE has conceived of the study. He has also participated in the design of the study, carried out the data collection, GIS and remote sensing based analysis of data, and performed the statistical analysis. MA and AL have participated in the sequence alignment of the draft manuscript. They also participated in its design and coordination, and helped to draft and edit the manuscript. All authors read and approved the final manuscript.

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Acknowledgements

The authors are grateful to Addis Ababa University and University of Gondar (Ethiopia) for their financial and operational support to conduct this research. This study was also made possible by a research grant awarded to the first author by Association of African Universities (AAUs). We also thank the local farmers in the study area for their cooperation and understanding during field work.

Competing interests

The authors declare that they have no competing interests.

Availability of data and materials

Not applicable.

Consent for publication

We have agreed to submit for Environmental Systems Research journal and approved the manuscript for submission.

Ethics approval and consent to participate

Not applicable.

Funding

No funding was received.

Publisher's Note

Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

Received: 25 May 2018 Accepted: 31 July 2018
Published online: 13 August 2018

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