Evaluation of soil erosion and sediment yield spatio-temporal pattern during 1990–2019

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ABSTRACT
Land use/land cover (LULC) change is an important measure to monitor the land degradation. This study focused on the Sebeya watershed located in the western province of Rwanda to evaluate the soil erosion and sediment yield spatio-temporal pattern from 1990 to 2019 occurred due to LULC change. Multi-temporal LULC maps of 1990–2019 and RUSLE model were used to achieve the objectives. The sediment yield was computed as a product of sediment delivery ratio (SDR) and the gross soil erosion. From the period 1990–2019, each year forestland decreased by 2.6%/yr, whereas the grassland, cropland, and settlement increased by 23.6%/yr, 14.5%/yr, and 81.6%/yr respectively. As a result, the mean soil erosion for 1990, 2000, 2010, and 2019 was 14.7, 28.3, 34, and 34.7 t ha⁻¹yr⁻¹ whereas the total sediment yield was 3254.15, 5029.8, 6814.56, and 9074.04 t yr⁻¹ respectively. Cropland was an exception to the other LULC classes because it has generated soil erosion in very high severity > 50 t ha⁻¹yr⁻¹, while this severity class showed an upward trend. The study demonstrated that LULC change has a significant impact on soil erosion and sediment yield and suggests proper land-use planning and encourages afforestation to arrest soil loss.

1. Introduction
Soil erosion is the leading cause of land degradation worldwide (Ebabu 2019; Kumar and Singh 2021). It alone contributes approximately 46% of the land degradation worldwide (Biggelaar et al. 2003), and is responsible for the 15–30 billion tons of the annual sediment transported by the world’s rivers into the ocean (Thomas et al. 2018).
Soil erosion was recognized as a problem in early civilizations (McCool and Renard 1990). Since earlier, it constrained the socioeconomic welfare, causing loss of soil fertility and productivity, increasing famine and malnutrition around the world. However, attention was given to the least developed countries because the latter struggle to feed an increasing population, coupled with human interference on natural forests, and inappropriate conservation measures accelerated land degradation in the form of soil erosion (McCool and Renard 1990; Issaka and Ashraf 2017; Ebabu 2019).

Rwanda’s lands are highly susceptible to soil erosion, about 78% of the total lands are exposed to soil erosion ranging from <5 to 100 t ha\(^{-1}\)yr\(^{-1}\) (Karamage et al. 2016). However, the country’s western region is the most susceptible (Kagabo et al. 2013; Kabirigi et al. 2017; Nyesheja et al. 2019; Byizigiro et al. 2020), and extensive conversion of natural forest into agriculture was indicated as the main leading cause (Karamage et al. 2016; Nambajimana et al. 2020). Over the last three decades, Rwanda’s land has been threatened by considerable LULC change and resulted into 64.5% of forest loss between 1990 and 2016 (Karamage et al. 2017).

As in the other parts of the world, the fastest change in LULC was due to increasing population pressure on forest resources, and agriculture land expansion to meet the graduated food demand, increasing demand for fuelwood and raw construction material, and the other anthropogenic influences (Mather and Needle 2000; Kobayashi 2004; Houghton 2012; Mesin Anteneh Wubie et al. 2016; Uwemeye et al. 2020). LULC change is consequently an important measure currently available to monitor pressures on ecosystems, biodiversity loss, global climate system, and land degradation (Meyer and Turner 1992; Riebsame et al. 1994).

In the context of the Sebeya watershed, its population density ranging from 1100 to 4850 inhabitants/km\(^2\) as per report of Water for Growth (W4GR 2018). Besides, agriculture accounts for half of the watershed geographic area, and it’s the main source of income for population livelihood as stated by the National Institute of Statistics of Rwanda (NISR 2018). In addition, about 47% of the total population lives in high poverty (W4GR 2018). Apart from that, Uwacu and Olalekan (2019) examined the physical and chemical parameters for water safety in the Sebeya Rivers, and the author reported the Turbidity level between 2330–3880 NTU, Total Suspended Solids (TSS) between 2455–1555 mg/l, and Chemical Oxygen Demand (COD) between 157–245 mg/l. These physical and chemical constituents concentration were far higher than the recommended World Health Organization (WHO 2004) standard and Rwanda Standard Board (RSB 2008). The above socioeconomic indications can accelerate the watershed’s land over-exploitation and can convert natural vegetation for agriculture purposes. As a consequence, the soil surface is left bared and vulnerable to soil erosion, and increase rivers sediment levels, while making unusable drinking water supply.

Several studies indicated a strong correlation between soil erosion severity and LULC change (Kavian et al. 2017; Gashaw et al. 2019; Özşahin and Eroğlu 2019). The authors concluded that LULC change is an important component to predict the past, actual, and future erosion at different spatial scales (Alkharabsheh et al. 2013; Esa et al. 2018; Thomas et al. 2018).
A wide range of models were developed to assess the rate of soil erosion (Wischmeier and Smith 1978; Knisel 1980; Beasley et al. 1982; Flanagan et al. 1995; Schmidt et al. 1999). Eventhough, the most popular models are the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978) and its revised version, the Revised Universal Soil Loss Equation (RUSLE) (Renard et al. 1991). The latter is widely adopted due to its simplicity, data availability, and compatible to GIS platform (Yang 2014; Thomas et al. 2018). Further, the model can better provide the relationship between LULC change and soil erosion (Phinzi and Ngetar 2019).

The RUSLE involved fixed parameters viz. topographical features (LS-factor), soil characteristics (K-factor) because these can remain constant over a long period, and controlled parameters viz. rainfall (R-factor), cover management (C-factor), and land management (P-factor) because they can control the severity of soil erosion (Renard et al. 1997). Still, the most controlled component is the cover management (C-factor) because it reflects the influence of types of vegetation, density, cropping, and management practices on soil erosion (Belayneh et al. 2019).

Tough the other variables are kept constant, the change in LULC changes the C-factor input variable and consequently affects the rate of soil erosion (Ozşahin and Eroğlu 2019; Phinzi and Ngetar 2019). Meanwhile, it’s worth appreciating the rate at which LULC change affects in space and time the soil erosion with better accuracy, and effectiveness, using the RUSLE model, GIS, and earth observation data (Sylla et al. 2012).

This study aimed to evaluate the soil erosion and sediment yield in space and time due to LULC change. Specifically, the study aimed: (1) to assess and map the LULC

Figure 1. Study area location map. Source: Author.
change during four sequential periods 1990, 2000, 2010, and 2019; (2) to estimate the
temporal trend of the soil erosion risk and determine the impact of LULC change on
soil erosion; and (3) to estimate the temporal trend of sediment yield at watershed
and sub-watersheds levels. The present study contributed to identifying the temporal
erosion patterns and soil erosion hotspot areas, and hence becomes an essential input
for planners and decision-makers to mitigate the soil erosion risk.

2. Materials and methods

2.1. Description of the study area

Sebeya watershed is positioned within 1°37’30” to 1°52’30” South Latitude and
29°16’00” to 29°28’30” East Longitude (Figure 1) and located on the western (Congo
River) side of the Congo-Nile divide (W4GR 2018). The catchment drains on
363.4 km² extended in four districts: Rutshuru, Rubavu, Ngororero, and Nyabihu. The
Sebeya River after flowing for a distance of 48 km from southern mountains in
Rutshuru district at 2660 meter above sea level (masl), it ends up into the Lake Kivu in
Rubavu town at 1470 masl.

Sebeya is a fourth-order river, and it features a dense network of water courses
with 6 streams of order three and 22 streams of order two. The watershed is located
in a heavy rainfall area receiving high annual mean rainfall above 1200 mm concen-
trated in two periods (February to May and September to December). The mean tem-
perature of 17°C tends to decrease due to high altitude (W4GR 2018).

Geologically, Sebeya is dominated by Granite base aquifer distributed throughout
the watershed except for the northern part that is dominated by basalt originated
from volcanic materials. Sebeya soil types are grouped into three categories: the
northern part is dominated by the young volcanic soil called mollic andosol (ANm)
with clay loam texture rich in organic matter (17.3%) with dark color, and covers
about 17% of the total area. The middle and eastern parts are dominated by less
developed humic Cambisols (CMu) with clay (light) texture which covers the most
prominent part, about 42% of the total area. Whereas the southern part is dominated
by humic Acrisols (ACu) with sandy clay which accounts 40.8%.

During the past three decades, the Sebeya watershed experienced a high forest dis-
appearance rate because of anthropological activities and high dependency on
Gishwati forest resources for local community livelihoods such as fuelwood, construc-
tion materials, cattle drenching, and agriculture (Uwemeye et al. 2020). This inter-
action between the local community and the natural environment caused agricultural
land to increase in space and time. Furthermore, since early there were poor farming
practices characterized by frequent tillage for seasonal crops and absence of sustain-
able anti-erosion control measures. Consequently, the study area underwent high to
extreme risk of rill and inter-rill erosion, and sedimentation, etc. This problem makes
the Sebeya watershed an appropriate case of study to evaluate the impact of LULC change on soil erosion and sediment yield.

2.2. Dataset

The Regional Centre for Mapping of Resources for Development (RCMRD) database that is available in the public domain (http://apps.rcmrd.org/landcoverviewer/) provided the historical LULC data for the years 1990, 2000, and 2010. The RCMRD closely works with the National Aeronautics and Space Agency (NASA) through the SERVIR Africa project and provides public domain high-quality land cover map of 10 years intervals starting from 1990 onwards. For each period, two classification schemes were generated: Scheme 1 depicting six classes namely forestland, cropland, grassland, settlement, wetlands, and other-land, and Scheme 2 illustrating the subclasses such as dense forest, moderate forest, sparse forest, woodland, closed grassland, open grassland, closed shrubland, open shrubland, perennial cropland, annual cropland, wetland, waterbody, settlement, and other-land. The land cover products are derived from the Landsat Thematic Mapper (Landsat 5) of 30 × 30 m of pixel size using the supervised Maximum Likelihood Classification method (MLC). The resulted products are deeply refined in the post-classification procedures, including pixel editing, filtering, and density slicing (RCMRD 2015). Furthermore, the refined land cover maps are subjected to accuracy checking using random points generated from the original Landsat imagery to produce highly transparent and consistent maps. These land cover maps are designed for providing baseline data for Agriculture, Forestry and Land Use (AFOLU) sector activities required for Green Houses Gases (GHGs) inventories that meet the International Panel on Climate Change (IPCC) requirement and cover nine Eastern and Southern Africa countries including Botswana, Ethiopia, Lesotho, Malawi, Namibia, Rwanda, Tanzania, Uganda, and Zambia. More detail about the historical land cover maps can be found from http://apps.rcmrd.org/landcoverviewer/. The updated 2019 LULC map was obtained using the Landsat 8 OLI delivered by the United States Geological Survey (USGS 2020) Earth Explorer (EE) database (https://earthexplorer.usgs.gov) of the respective path and the row of 173, and 061 acquired on August 15, 2019.

The monthly rainfall data (1990–2019) have been obtained from the East African grids Climate Hazards Group InfraRed Precipitation (CHIRPS). CHIRPS incorporates 0.05° × 0.05° resolution satellite imagery (Funk et al. 2015). A region of interest (ROI) was extracted, and the average annual rainfall was computed to find the 30 years of yearly rainfall data. The latter was further used to calculate the long-term mean annual rainfall (P) of the thirteen grids covering the entire Sebeya watershed. Topographic features were extracted from the one arc-second Shuttle Radar and Topographic Mission (SRTM) Digital Elevation Model (DEM) Version 3 of tile S02_E29 with a pixel size of about 30 × 30 m acquired from USGS Earth Explorer (http://earthexplorer.usgs.gov). Soil properties were extracted from the Harmonized World Soil Database (HWSD) version 1.2 of the spatial resolution of 30 arc-second (FAO 2013). The HWSD is a compilation of different databases and have 221 million grid cells that cover the entire globe. Each grid is linked to the attribute database for
soil parameters queries such as soil unit name, soil depth, sand, silt and clay percentage, texture classes, organic carbon percentage, drainage class, pH, bulk density, water storage capacity, gypsum, cation exchange capacity of the soil, and granulometry, etc. Since the datasets were provided in different spatial resolutions, they were resampled to a common spatial resolution of $30 \times 30$ m and projected to the same reference Universal Transverse Mercator Zone 35S, WGS Datum to generate high-resolution periodical erosion maps.

2.3. Data analysis

2.3.1. LULC classification

The RCMRD Land cover maps of 1990, 2000, and 2010 scheme 1 were adopted because classes illustrated in this classification scheme were more representative to elucidate the impact of LULC change on soil erosion (Nambajimana et al. 2020). The Landsat 8 image was used to prepare the land cover map of 2019. In general, five classes were identified referring to RCMRD land cover classes (forestland, grassland, cropland, settlement, and waterbody) processed in ENVI software version 5.3. The generated classification product was subjected to the comparison to the ground reference, commonly termed as accuracy assessment (Mather 2004). Total 40 points were randomly sampled for each class on the original Landsat image using the sampling tool located in the Data management toolset of ArcGIS software. These points given in the shapefile were converted to KML format compatible with the Google Earth (GE) program. Then after the land cover classification accuracy verification was applied by overlaying the classified image on the Very High-Resolution Satellite Images (VHRSI) from GE. This technique was preferred because it was successfully proven to be reliable in the recent comparative studies (Karamage et al. 2016; Kayiranga et al. 2016; Qamer et al. 2016; Akinyemi 2017; Gashaw et al. 2019). The result provides the overall accuracy, producer’s accuracy to measure omission error, user’s accuracy in measuring commission error, and Kappa coefficient to measure the overall agreement between the classified thematic map and the ground reference data (Story and Congalton 1986; Congalton 1991).

2.3.2. Soil loss estimation

The RUSLE is a popular model applied in several studies to assess the rill and inter-rill soil erosion (Kulimushi et al. 2021; Kumar and Singh 2021).

RUSLE assesses the soil loss as a direct product of five erosion factors (Equation (1)), including rainfall erosivity (R-factor), soil erodibility (K-factor), slope length, and steepness (LS-factor), cover management (C-factor), and support practice (P-factor) (Renard et al. 1997).

$$A = R \times K \times LS \times C \times P$$  

$A =$ annual mean soil loss (t ha$^{-1}$ per year), $R =$ Rainfall factor (MJ ha$^{-1}$ per year), $K =$ Soil erodibility factor (t ha h MJ$^{-1}$ mm$^{-1}$), $LS =$ Slope length and steepness (dimensionless), $C =$ Cover-management factor (dimensionless), and $P =$ Land management factor (dimensionless) (Renard et al. 1997).
The quantitative estimation of soil erosion was computed four times in 1990, 2000, 2010, and 2019. The R, K, LS, and P factors were kept constant whereas the C-factor was periodically changed as the LULC changed. Erosion hazard maps were reclassified into five severity classes: very low (0–5 t/ha/year), low (5–15 t/ha/year); moderate (15–30 t/ha/year), high (30–50 t/ha/year), very high (>50 t/ha/year) adapted from (Esa et al. 2018; Kidane et al. 2019).

2.3.2.1. Estimation of the rainfall erosivity (R-factor). Rainfall erosivity is a multifaceted process in which the amount, intensity, energy, duration, pattern, size of raindrops of rainfall, and associated runoff exerts influence. In tropical regions, Rainfall erosivity is estimated using Equation (2) proposed by Lo et al. (1987) and employed in (Nambajimana et al. 2020; Kulimushi et al. 2021).

\[
R = 38.46 + (3.48 \times P)
\]  

Equation (2)

P is the long term mean annual rainfalls for 30 years (mm).

Since R-factor is supposed to remain constant over time, a unique long-term mean annual rainfall (P) for the 30 years was calculated. Then, P was used as input in Equation (2) to calculate the erosivity factor in the raster calculator. The generated R-factor map was converted to point and interpolated using the Inverse Distance Weight (IDW). This exercise makes the erosivity values more reliable on the whole surface, and the IDW technique compared to Kriging; provides the low error for mapping the rainfall data as reported (Panditharathne et al. 2019).

2.3.2.2. Estimation of slope length and steepness (LS-factor). The LS-factor represents the overall contribution of topographic features such as length and steepness of the slope on soil erosion (Wischmeier and Smith 1978). LS-factor is a reference ratio of soil loss under standard slope length of 22.13 m and steepness of 0.09 rad about (5.14° or 9%) of a USLE plot unit (Renard et al. 1997). The topography accelerates progressively the runoff downslope and increases the erosivity of runoff. The DEM used to prepare the LS-factor was extracted from the SRTM. This DEM was subjected to geometric and hydrologic corrections and was hence used to generate flow direction, flow accumulation, and slope gradient. With the help of the above inputs, L-factor was computed to the relationship proposed by Desmet and Govers (1996) (Equation (3)) and S-factor was calculated after McCool et al. (1987) (Equation (4)). Then, in the Spatial Analyst tool of ArcGIS software, the LS factor was calculated as a product of L and S factors.

\[
L_{i, j} = \frac{\left( A_{i, j}, \text{in} + D^2 \right)^{m+1}}{x_{i, j, \text{in}} \times D^{m+2}} \times 22.13^m \quad \text{where} \quad m = \frac{\beta}{1 + \beta} \quad \text{and} \\
\beta = \frac{\sin \theta}{3(\sin \theta)^{0.8} + 0.56}
\]  

Equation (3)
\[
S = \begin{cases} 
10.8 \times \sin \theta_{(i,j)} + 0.3, & \tan \theta_{(i,j)} < 9\% \\
10.8 \times \sin \theta_{(i,j)} + 0.3, & \tan \theta_{(i,j)} < 9\% 
\end{cases}
\]

where \(A_{(i,j)in}\) computed in GIS is replaced by the flow accumulation, which indicates the accumulated upslope contributing area per unit cell; \(D\) is the grid cell size in (meters); \(m\) is a variable slope length exponent related to the \(\beta\) ratio of rill (caused by flow) to inter rill erosion (caused by raindrop) (McCool et al. 1989), \(x_{i,j} = (\sin a_{i,j} + \cos a_{i,j})\); \(\beta\) = ratio of rill to interrill erosion; \(\theta\) is the slope angle in degree.

### 2.3.2.3. Estimation of the erodibility (K-factor)

The soil erodibility value refers to the influence of soil properties on soil loss during storm events in the cultivated areas (Wischmeier and Smith 1978). K-factor shows the level of susceptibility of a soil type to detachment and transport during intense rainfall. The values closed to 0 are the least susceptible, the range between 0.2 and 0.4 is moderate and values greater than 0.4 are most susceptible (Kulimushi et al. 2021).

The K-factor is computed using the following Equation (5):

\[
K = 2.1 \times 10^{-6} \times M^{1.4} \times (12 - OM) + 0.0325 \times (P - 2) + 0.025 \times (S - 3)
\]

where \(M = (\%\text{ silt} + \%\text{ very fine sand}) (100\%\text{ clay})\); \(OM = \) the percentage of organic matter; \(P = \) Permeability class; and \(S = \) structure class.

Generally, the K-factor is the function of many soil physical parameters obtained from soil samples analysis, such as particle size, organic matter, permeability, and structure classes (Equation (5)). However, many authors proved the usefulness of the secondary data from various regional and global sources to assess the soil erodibility and its potential impact on soil erosion (Robert et al. 2015; Chadli 2016; Balabathina et al. 2019). These authors concluded that two of the four physical parameters (texture classes or particle size and the organic matter content) can be used to generate soil erodibility. This is because the smaller particle increases the cohesiveness of soil and larger particle size increase surface permeability. In addition, the mass of organic matter increases the resistance of soil and reduces the erosivity effect on soil particles.

Consequently, the present investigation used the relevant soil parameters derived from the Harmonized World Soil Database v1.21 (FAO et al. 2013). Following Stone and Hilborn’s (2000) factsheet (Table A1), K-factor was estimated using USDA soil texture classification and organic matter percent (%SOM). In fact %SOM was not available in the database, this parameter was therefore derived from the available organic carbon content (%SOC) multiplied by a conversion factor of 1.724 termed as Van Bemmelen factor (Griffin and Edward 2019). The observed K-factor value given in American system units (U.S. units) is converted to the International System Units (S.I. Units) using 0.1317 value to be expressed in tons ha MJ\(^{-1}\)mm\(^{-1}\) (Renard et al. 1997).
2.3.2.4. Estimation of cover management (C-factor). The C-factor refers to cropping and crop management practices’ effect on soil erosion rate in agricultural lands (Wischmeier and Smith 1978; Renard et al. 1997). It might be the most important factor to predict the past, actual, and future soil erosion severity because of its dynamism in short term (Alkharabsheh et al. 2013; Yang 2014; Kavian et al. 2017; Gashaw et al. 2019; Özsahin and Ergolu 2019; Phinzi and Ngetar 2019). The C-factor is dimensionless and ranges from near 0 (well protected) to 1 (barren soils before plants grow).

This factor was prepared from the corresponding land cover maps and fixed values were assigned to every LULC class (Kidane et al. 2019; Panditharathne et al. 2019). The work allocated 0 for settlement, 0.01 for grassland, 0.03 for forestland, and 0.21 for cropland referred to (Thakuri et al. 2019; Byizigiro et al. 2020). Hence, four C-factor raster grids were derived for 1990, 2000, 2010, and 2019 to assess the impact of LULC change on soil erosion.

2.3.2.5. Estimation of support practice (P-factor). This factor reflects the impact of supporting practices (contours, terraces, strip cropping, and tillage) on erosion rate with tillage up and downslope (Renard et al. 1997). This factor can be determined by different conservation measures if they are applied (Belayneh et al. 2019). However, there was an absence of anti-erosive measures implemented within the timeframe (W4GR 2018). Consequently, a fixed P-factor value of 0.75 was considered referred to (Karamage et al. 2016) which signifies unsustainable conservation measures implemented.

2.3.3. Estimation of the sediment delivery ratio (SDR)

Sediment delivery ratio (SDR) is defined as a proportion of gross erosion delivered to the outlet of the watershed area in a given period (Kidane et al. 2019). Several approaches are used to estimate the SDR which depends on data availability and many factors such as watershed area, channel slope, stream density, LULC, sediment source, rainfall-runoff, still they can all yield the same outcome (Ouyang and Bartholic 1997). The work adopted a procedure proposed by Williams and Berndt (1972) (Equation (6)). According to Williams’s method, topographical features such as stream height ratio and stream length influence sediment transportation within the watershed. This method was also used by other similar studies (Panditharathne et al. 2019; Kidane et al. 2019). The method used the average channel slope to derive the SDR (Equation (6)).

\[
SDR = 0.627 \times (SLP)^{0.403} \quad \text{where} \quad SLP = \frac{\Delta Ch}{Lc} \tag{6}
\]

SDR = stream slope expressed in percent; \(\Delta Ch\) is the channel divide and outlet; \(Lc\) is the distance between the same two points parallel to the main channel. Hec GeoHMS was used to compute the stream slopes after mapping the flow direction, accumulation, and drainage network. SDR was finally computed in the raster calculator using Equation (7):
2.3.4. Estimation of the sediment yield (SY)
Sediment Yield (SY) consists of deposition of the eroded fraction detached and transported by runoff (Ouyang and Bartholic 1997). Sediment yield (Equation (8)) is a function of sediment delivery ratio and gross erosion. These variables are commonly used to estimate the SY because this parameter is not available as a direct measurement and because of the lack of measurement structure at the river’s mouth (Ouyang and Bartholic 1997; Kidane et al. 2019). Four SY raster grids were derived for 1990, 2000, 2010, and 2019 assuming that the change in LULC impacts the sediment yield in space and time. The spatio-temporal variation of SY was hence attributed to the spatio-temporal variation of LULC. A Raster calculator tool of ArcGIS software was used to compute four periodical SY maps using Equation (8).

\[
SY = SDR \times C2
\]

\[
SDR = 0.627 \times SLP^{0.403}
\]

SY: Sediment yield (t ha\(^{-1}\) y\(^{-1}\)), SDR: Sediment delivery ratio, and A: annual mean soil loss t ha\(^{-1}\) y\(^{-1}\).

3. Results

3.1. Overview of the sebeya LULC change
The overall accuracies of land cover maps for 1990, 2000, and 2010 were 82.2%, 82.7%, and 81.3% and the Kappa coefficients were 0.75, 0.76, and 0.74 respectively (http://apps.rcmrd.org/landcoverviewer/). Whereas for 2019, the overall accuracy and Kappa coefficient were 82.7% and 0.8 respectively (Table A2). The overview of Sebeya LULC change are given in Tables 1 and 2 and displayed in Figure 2.

Considerable change of LULC was detected during the period under investigation. In the 1990s forestland was accounted for 87.6% of the total watershed area. Within a decade 1990–2000, forestland lost about 36.7%, i.e. it comprised for 51% in 2000. In the next decade 2000–2010, forestland lost about 13.8% and covered 37.1% in 2010. During the last decade 2010–2019, forestland lost again 19% of its total area and actually accounts for 18.1% in 2019 Table 1. In general, between 1990 and 2019 forestland lost 69.5% of its initial total coverage i.e. from 87.6% (31979.4 ha) to 18.1% (6603 ha) (Table 1). Karamage et al. (2017) reported 64.5% of the forest loss during the period 1990–2016, and the author attributed this drastic change of forest to the rapidly growing population, and substistence agriculture for livelihood.

The dependence on natural forest resources has caused a major change in cropland within the period of study (NISR 2018). In the 1990s cropland accounted for 7.8% . This little surface was enough to feed a few humans who’s settled on 0.2% of the watershed area. Within a decade , cropland gained about 21.6% and accounted for 29.4% in 2000. This period corresponds to the wars and genocide against Tutsi that occurred in 1994 when severe forest resources were overexploited (NISR 2018). Moreover, between 2000 and 2010 cropland gained about 8.6% and comprised
approximately 38% in 2010. In the last decade, cropland gained only 4%, still is the predominant land use category because it moved from 7.8% (2850.2 ha) in 1990 to 42% (15316.7 ha) in 2019 (Table 1).

Grassland has also increased over time, in the 1990s grassland covered about 4.4% of the total area. Within a decade 1990–2000, grassland experienced considerable change and comprised for 19% of the total area in 2000. In the next decade, i.e. between 2000 and 2010, grassland moved to 23.9% while in the last decade, it increased by 11.4% to cover 35.3% in 2019. During the past three decades grassland increased about 30.9% compare to its 1990s total coverage i.e. from 4.4% (1594 ha) to 35.3% (12886.4 ha) in 1990 and 2019 respectively. The increase of grassland including (grass, shrub, and woodlands) was due to some ongoing initiative of forest rehabilitation as stated by Uwemeye et al. (2020).

Settlement has also undergone an increasing change within the period of study. Settlement accounts for 0.2% in 1990, 0.6% in 2000, 1.1% in 2010 and 4.6% in 2019 while waterbody covered 0.003%, 0.054%, 0.003% and 0.002% in 1990, 2000, 2010 and 2019 respectively.

The result illustrated in Table 2 revealed that from 1990 to 2019 cropland increased about 12466.5 ha equivalent to the annual rate of about 415.5 ha/year (14.5%). Similar to grassland increased about 11292.4 ha approximately to an annual increase of 376.4 ha/year (23.6%), and settlement also increased 1617 ha corresponding to 53.9 ha/year (81.6%) as annual increasing rate. Contrary to forestland which loss rate was estimated at 845.9 ha/year (2.6%) corresponding to a total loss of about 25376.3 ha. The surface covered by waterbody also experienced a temporal change which might be due to the variation of flooding zones at satellite acquisition.

### Table 1. Annual distribution of LULC classes between 1990 and 2019.

| LULC    | 1990 Area ha | 1990 % | 2000 Area ha | 2000 % | 2010 Area ha | 2010 % | 2019 Area ha | 2019 % |
|---------|--------------|--------|--------------|--------|--------------|--------|--------------|--------|
| Forestland | 31979.4     | 87.6   | 18579.8      | 50.9   | 13527.3      | 37.1   | 6603.1       | 18.1   |
| Grassland | 1594.0       | 4.4    | 6948.0       | 19.0   | 8705.5       | 23.9   | 12886.4      | 35.3   |
| Croplands | 2850.2       | 7.8    | 10724.7      | 29.4   | 13856.6      | 38.0   | 15316.7      | 42.0   |
| Settlement | 66.0         | 0.2    | 217.9        | 0.6    | 399.8        | 1.1    | 1683.7       | 4.6    |
| Waterbody | 0.9          | 0.0    | 20.0         | 0.1    | 1.3          | 0.0    | 0.8          | 0.0    |
| Total    | 36490.5      | 100    | 36490.5      | 100    | 36490.5      | 100    | 36490.5      | 100    |

3.2. Development of erosion factors database

3.2.1. Rainfall erosivity (R-factor)

Table 3 presents the sum of rainfall, mean annual rainfall, and erosivity factor value per grid. Whereas Figure 3a shows the spatial distribution of the erosivity value throughout the study area. The calculated time-series mean monthly rainfall per grid/year indicates a range from 1077 mm/y to 1302.2 mm/y with a mean and standard deviation of 1221 mm/y and 70 mm/y respectively. The observed trend of the mean annual rainfall indicated a slightly increasing trend. Considering the respective periods under investigation, i.e. in 1990, 2000, 2010, and 2019, the mean annual rainfall was 1371.1 mm/y, 996.69 mm/y, 1427.83 mm/y, and 1155.1 mm/y respectively (Figure
Moreover, it has been observed two dominated rainfall periods (February to May and September to December) recorded about 84% of the total mean rainfall (Figure A2).

Erosivity result displayed in Figure 3(a) obtained from the IDW technique indicated that the Sebeya watershed is characterized by high long-term erosivity with

| LULC changes | 1990–2000 | % | 2000–2010 | % | 2010–2019 | % | 1990–2019 | % | Annual change |
|--------------|-----------|---|-----------|---|-----------|---|-----------|---|--------------|
| Forestland   | −13399.6  | −36.7 | −5052.6   | −13.8 | −6924.2   | −19.0 | −25376.3 | −69.5 | −845.9       |
| Grassland    | 5354.0    | 14.7 | 1757.5    | 4.8  | 4180.9    | 11.5 | 11292.4  | 30.9 | 376.4        |
| Cropland     | 7874.6    | 21.6 | 3131.8    | 8.6  | 1460.1    | 4.0  | 12466.5  | 34.2 | 415.5        |
| Settlement   | 151.9     | 0.4  | 181.9     | 0.5  | 1283.9    | 3.5  | 1617.6   | 4.4  | 53.9         |
| Waterbody    | 19.1      | 0.1  | −18.7     | −0.1 | −0.6      | 0.0  | −0.2     | 0.0  | −0.0         |

Figure 2. Land use/land cover mapping for: (a) 1990; (b) 2000; (c) 2010; and (d) 2019. Source: Author.
value ranging from 3656.9 to 4637.6 MJ mm ha$^{-1}$ h$^{-1}$ yr$^{-1}$ with a mean erosivity and standard deviation of 4339.5 MJ mm ha$^{-1}$ h$^{-1}$ yr$^{-1}$ and 181.8 MJ mm ha$^{-1}$ h$^{-1}$ yr$^{-1}$ respectively. About 90% of the entire watershed has an erosivity greater than 4000 MJ mm ha$^{-1}$ h$^{-1}$ yr$^{-1}$, whereas 63.8% of the total area indicated an erosivity range from 4300 to 4637.6 MJ mm ha$^{-1}$ h$^{-1}$ yr$^{-1}$. Despite the spatial variation of the erosivity factor, the entire watershed is highly erosive as the R-factor is by far greater than the global average R-factor of 2000 MJ mm ha$^{-1}$ h$^{-1}$ yr$^{-1}$.

### 3.2.2. Slope length and steepness (LS-factor)

High LS factor is expected in the catchment due to high value of the contributing inputs. LS-factor result is shown in Figure 3(b). The result indicated the range from 0 in the lower slope to 1002 in steeper slope with a mean and the standard deviation of 4.8 and 5.6 respectively. It has been observed that LS-factor value increases with the elevation from 2000 and slope above 15°. Besides, about 97.7% of the total watershed area has LS-factor between 0 and 10. The result revealed that high LS-factor predominates the southern, and middle parts to eastern highlands and those areas are characterized by very steep slopes. In these parts of the watershed, runoff energy might be high due to high LS-factor and revealed a potential soil erosion (Phinzi and Ngetar 2019).

### 3.2.3. Erodibility factor (K-factor)

The results presented in Table 4 and Figure 3(c) revealed a variation of the K-factor due to texture classes. The K-factor ranged from 0.018 to 0.037 tons ha MJ$^{-1}$ mm$^{-1}$ with a mean of 0.025 tons ha MJ$^{-1}$ mm$^{-1}$. The same range was recently reported in Rwanda by Nambajimana et al. (2020).

The result noticed that K-factor greater than 0.028 tons ha MJ$^{-1}$ mm$^{-1}$ is concentrated in the northern part of the watersheds, this part is characterized by the basalt layer originated from volcanic materials and retrieved from low-gradient footslope (Batjes 2007). Moreover, K-factor ranging between 0.018 and 0.028 tons ha MJ$^{-1}$ mm$^{-1}$ covers about 42% of the total watershed area. This is found toward the eastern and middle parts with Cambisols as the predominant soil groups. The southern part has a low K-factor 0.018 tons ha MJ$^{-1}$ mm$^{-1}$ and this part is dominated by humic Acrisols, granite, and pegmatite geological rocks (W4GR 2018).
Figure 3. (a) Erosivity factor, (b) Slope length and steepness factor, (c) Erodibility factor, (d) Cover management in 1990, (e) Cover management in 2000, (f) Cover management in 2010, (g) Cover management in 2019. Source: Author.
3.2.4. Cover management factor (C-factor)
The dimensionless C-factor values ranged from 0, 0.01, and 0.03 to 0.21 for settlement, grassland, forestland, and cropland respectively were adopted from Thakuri et al. (2019) and Byizigiro et al. (2020). The C-factor maps are shown in Figure 3(d–g). A low C-factor value is likely to generate lower soil erosion while the rate of soil erosion increase by increasing in C-factor value (Özşahin and Eroğlu 2019; Kulimushi et al. 2021).

The observed considerable change in LULC is likely to impact and expose the watershed to severe soil erosion due to the drastic increase of high C-factor in space and time. This assumption was confirmed by several studies (Alkharabsheh et al. 2013; Esa et al. 2018; Balabathina et al. 2019). The authors reported a greater share of the total soil loss generated in cropland because the seasonal variation in crops production left the soil surface bared and uncovered (Panditharathne et al. 2019). Which fact can increase the surface runoff due to the absence of vegetation while the latter plays a role in dissipating the rainfall energy before it impacts the soil. As a consequence, the loss of vegetation makes the surface vulnerable to soil erosion coupled to marginal topography it also increases the accumulation of sediments in reservoirs.

3.2.5. Support practice (P-factor)
Proper conservation measures are the alternative to counter the high occurrence of soil erosion (Pimentel and Burgess 2013). However, the anti-erosion measure is not implemented in the watershed and is still under the project phase (W4GR 2018). The farming practices are still dominated by rudimentary agriculture methods characterized by frequent and regular intensive tillage with ineffective erosion control measures (W4GR 2018). Consequently, a static P-factor of 0.75 was allocated (Karamage et al. 2016). This signifies inexistent conservation measures and a high risk of soil erosion.

3.3. The trend of soil erosion risk
3.3.1. Distribution of soil erosion
The soil erosion rate and the total soil loss are presented in Figure 4(a,b). The result revealed a temporal variation of the soil erosion rate along with the total soil loss within the period under investigation. The soil erosion rate and standard deviation were 14.7 ± 24.9, 28.3 ± 44.7, 34 ± 66.2, and 34.7 ± 67.7 t ha⁻¹ yr⁻¹ in 1990, 2000, 2010, and 2019 respectively (Figure 4(a)). The actual erosion rate is greater than the tolerable rate (10 t ha⁻¹ yr⁻¹) in the tropical ecosystem. The estimated total soil loss were 219468 tons yr⁻¹; 421323.6 tons yr⁻¹; 506412.6 tons yr⁻¹ and 516708.7 tons yr⁻¹ for 1990, 2000, 2010, and 2019, respectively (Figure 4(b)).

3.3.2. Soil erosion severity classes
The area and contribution to the total soil loss of each severity class are presented in Table 5, while Figure 5 displays the spatial distribution of the soil erosion severity classes in different periods. Initially, the watershed was exposed to negligible (very low and low) soil erosion at a rate of 70% and these classes shared about 37.1% of the total soil loss in 1990. In the same period, moderate soil loss (15–30 t ha⁻¹ yr⁻¹)
accounted for 24.2% and was responsible for 32.12% of the total soil loss. Whereas high (30–50 t ha\(^{-1}\) yr\(^{-1}\)) and very high (>50 t ha\(^{-1}\) yr\(^{-1}\)) severity classes comprised for only 1.88% and 3.47% of the total area and contributed by the same order 4.83% and 25.9% of the total estimated soil loss.

In the next decades, a significant change in soil loss rate was observed. For instance, in 2000 and 2010 low erosion severity accounted for 57.3% and 55.6% and contributed to the total soil loss by 12.6% and 10.2% respectively (Table 5). Whereas the result noticed a spatial increase of the soil loss higher than 30 t ha\(^{-1}\) yr\(^{-1}\). These classes increased to 24.76% and 29.05% in 2000 and 2010 with the respective contribution to the total soil loss of 74.75% and 80.65%. The actual erosion status indicated that about 32% of the watershed area is exposed to an erosion rate higher than 30 t ha\(^{-1}\) yr\(^{-1}\) and these classes contribute by 84.7% to the total soil loss in 2019. The 56.1% of the watershed is still exposed to low soil loss but contributes too little about 8% to the total soil loss in 2019. During the past three decades i.e. between 1990 and 2019, the low severity class lost about 14.28% of its initial coverage, and moderate severity lost about 12.55%. Conversely to high and very high gained both about 26.83% compared to their initial coverage and they contributed a lot to the total soil loss.

3.3.3. Soil erosion rate under different LULC categories

Table 6 revealed that cropland was the only class to generate high and very high soil erosion rates. The mean soil erosion in cropland ranged from 44.4 t ha\(^{-1}\) yr\(^{-1}\), 62 t ha\(^{-1}\) yr\(^{-1}\), 67.9 t ha\(^{-1}\) yr\(^{-1}\), to 67.8 t ha\(^{-1}\) yr\(^{-1}\) in 1990, 2000, 2010, and 2019 and respectively contributed to the total soil loss by 23.1%, 64.5%, 76.17%, and 82.3% (Table 6). The observed high erosion rate in cropland was attributed to socioeconomic reasons, the farming practices, and the absence of support practice. As stated Özşahin and Eroğlu (2019) good support practice improves the C-
factor. Furthermore, it’s due to the forest removal leaving the soil surface bared and impact on soil erosion rate occurred in cropland (Azimi et al. 2019).

Around 87.6% of the total watersheds area was covered by forestland in 1990 and this land cover generated the negligible erosion rate of 12.5 t ha⁻¹ yr⁻¹ and contributed by 74.6% to the total annual soil loss. However, the forest disturbance by local community occurred in the following decades has increased the soil erosion rate in forestland. In 2000, 2010, and 2019 the soil erosion rate in forestland was 16.91 t ha⁻¹ yr⁻¹, 16.45 t ha⁻¹ yr⁻¹, and 19.41 t ha⁻¹ yr⁻¹ respectively and contributed to the total soil loss by 30.5%, 17.8%, and 9.9% respectively.

The temporal variation of soil erosion generated in grassland indicated an average from 7.25 t ha⁻¹ yr⁻¹, 7.25 t ha⁻¹ yr⁻¹, and 8.38 t ha⁻¹ yr⁻¹, to 7.26 t ha⁻¹ yr⁻¹ from 1990, 2000, and 2010 to 2019, respectively. In the same order shared 2.1%, 4.8%, 5.8%, and 7.4% of the total soil loss. Grassland’s contribution to the total soil loss is attributed to the considerable increase of its spatial coverage. The settlement has undergone a very low soil erosion rate ranged from 1.52 t ha⁻¹ yr⁻¹, 1.83 t ha⁻¹ yr⁻¹, and 1.3 t ha⁻¹ yr⁻¹, to 1.56 t ha⁻¹ yr⁻¹ from 1990, 2000, and 2010 to 2019, respectively.

### 3.3.4. Temporal change of soil erosion rate under different LULC categories

Cropland and forestland conversely to grassland and settlement had experienced a temporal change in the soil erosion rate (Figure 6). The soil erosion rate in cropland increased about 17.55 t ha⁻¹ yr⁻¹ between 1990 and 2000 and increased about 6 t ha⁻¹ yr⁻¹ between 2000 and 2010. However, during the last decade, the soil erosion rate continued to increase.
rate in cropland stabilized, it experienced a slight decrease of 0.16 t ha\(^{-1}\) yr\(^{-1}\) between 2010 and 2019 (Figure 6). During the past three decades i.e. between 1990 and 2019 the soil erosion rate generated by cropland increased about 23.34 t ha\(^{-1}\) yr\(^{-1}\) compared to its initial status. This was due to extensive increase of agriculture of subsistence, and together with inadequate conservation measures (Balabathina et al. 2019). However, in the meantime, grassland and settlement experienced an insignificant change in soil erosion during the period under investigation.

The soil erosion rate in forestland has experienced a temporal change as result reported. The soil erosion rate generated in the forestland increased about 4.37 t ha\(^{-1}\) yr\(^{-1}\) between 1990 and 2000 but it slightly decreased about 0.46 t ha\(^{-1}\) yr\(^{-1}\) between 2000 and 2010. Furthermore, between 2010 and 2019 the soil erosion rate in forestland increased about 2.95 t ha\(^{-1}\) yr\(^{-1}\). During the past three decades, the soil erosion rate generated in the forestland indicated an increase of 6.8 t ha\(^{-1}\) yr\(^{-1}\) as illustrated in Figure 6. This was due to the forest disturbance that occurred in the past
decades that increasing the vulnerability to soil erosion. Furthermore, it might be due to the characteristics of the geo-environmental settings of forestland including marginal topography with steep slopes $> 30^\circ$, erosivity higher than 4000 MJ mm ha$^{-1}$ h$^{-1}$ yr$^{-1}$ etc.

### 3.4. Temporal trend of sediment yield

The sediment delivery ratio was ranged from 0 to 0.31 (31%). This means, up to 31% of the total soil loss is delivered in the drainage system. In other words, the eroded material transported through runoff is converted to sediment yield up to 31% while a fraction of $\geq 69\%$ is deposited in downslope areas (Ouyang and Bartholic 1997).

The mean sediment yield and total sediment yield are presented in Figure 7(a,b) while Figure 8 depicts the spatial-temporal distribution of the mean sediment yield.

The result indicated a temporal variation of the mean sediment yield and total sediment during the investigated periods. The mean sediment yield and standard deviation moved from $4.68 \pm 14.1$, $7.26 \pm 20.35$, and $9.8 \pm 50.96$, to $13 \pm 65.46$ t ha$^{-1}$ yr$^{-1}$ from 1990, 2000, and 2010 to 2019 respectively. Furthermore, the total sediment yield moved from 3254.15 tons yr$^{-1}$, 5029.8 tons yr$^{-1}$, and 6814.56 tons yr$^{-1}$ to 9074.04 tons yr$^{-1}$ from 1990, 2000, and 2010 to 2019 respectively (Figure 7(a,b)). The
temporal change of mean sediment yield at the sub-watershed level was also noticed and presented in Table 7.

Initially, the mean sediment yield at the sub-watershed scale ranged from 1.08 to 4.42 t ha\(^{-1}\) yr\(^{-1}\) for the Sebeya downstream and upstream sub-watersheds respectively. Within a decade the range moved from 3.64 to 9.53 t ha\(^{-1}\) yr\(^{-1}\) in 2000 for the Sebeya downstream and upstream sub-watersheds respectively. Moreover, in 2010 the mean sediment yield ranged from 4.87 to 19.21 t ha\(^{-1}\) yr\(^{-1}\) for the Sebeya upstream and the Karambo sub-watersheds respectively. Whereas in 2019 the mean sediment yield at sub-watersheds ranges from 8.58 to 34.62 t ha\(^{-1}\) yr\(^{-1}\) for the Sebeya upstream and the Karambo sub-watersheds respectively.

The contribution of the sub-watersheds to the total sediment loading in rivers indicated that Sebeya downstream sub-watershed had the least mean sediment yield in 1990, but it contributed by 26.07% to the total sediment yield following by Pfunda, Karambo, Upstream, and Bihongoro with a respective initial share of 22.94%, 20.09%, 17.8% and 13.9% in 1990.

The result further indicated that out of 9074 tons of sediment yield produced in 2019, about 42.36% come from the middle part representing Karambo sub-watersheds. Whereas Downstream, Upstream, Pfunda, and Bihongoro shared about 19.83%, 19.19%, 9.93%, and 8.69%, respectively (Table 7). As consequence of the high sediment load into the Sebeya River is the increase in drinking water treatment cost because the sedimentation of the eroded material makes water unusable for drinking and irrigation. Furthermore, it exposes downstream areas to flooding which impacts infrastructures and residential properties (W4GR 2018).

4. Discussions
4.1. LULC change

During the period under investigation the watershed have undergone a considerable change of LULC observed through forest clearance on behalf of the others land uses. The natural forest resources experienced high deforestation at a rate of 2.6% of loss per year equivalent to 845 ha/year. This deforestation mainly profits to agriculture
Figure 8. (a) Sediment delivery ratio, (b) Sub-watersheds map, (c) Sediment yield in 1990, (d) Sediment yield in 2000, e. Sediment yield in 2010 and f. Sediment yield in 2019. Source: Author.
expansion growing at a rate of 14.5% each year equivalent to 415 ha of gain per year. This change is driven by exponential food demand from population growth that poses a challenge to agriculture to meet the changing population’s food demand and hence pushing farmers to discover new lands toward protected areas (Dyson 1999).

For instance, the density of the population in the Sebeya varies from 1100 to 4,850 inhabitants/km², and about half of the population are poor and depend on agriculture for their livelihoods (NISR 2018; W4GR 2018). Consequently, the socioeconomic conditions led to land scarcity and pressure shifting to natural forest and encroachment of the other hilly areas (Kayiranga et al. 2016; Nyesheja et al. 2019; Uwemeye et al. 2020).

Between 1990 and 2000 the Sebeya’s forest lost about 36.7% while cropland gained about 21.6% of its initial spatial coverage. This is because during and after the 1994 Genocide against the Tutsi, there was a shortage of lands to resettle returnees and internally displaced people. Consequently, forests were encroached for anthropogenic activities agriculture and resulted in considerable loss of forestland (Clay 2019; Uwemeye et al. 2020). NISR (2018) also reported a considerable change in LULC occurred in Rwanda between 1990 and 2000 due war and instability, and as a consequence, anthropogenic activities started taking place in natural forests.

The land cover map of 2019 indicated potential afforestation especially from the eastern highlands of the watershed representing the western part of Gishwati National Park. This afforestation was attributed to the government law enforcement against deforestation (Ordway 2015). Different stakeholders were involved to protect and rehabilitate the forest, these include the government institutions, local community, non-government organizations (NGOs), and the surrounding population (Uwemeye et al. 2020). Furthermore, there was the implementation of a project known as Landscape Approach to Forest Restoration and Conservation (LAFREC) implemented to enhance the climate-resilient livelihoods and environmental services through forest rehabilitation to later consider Gishwati forest Reserve as a National Park in 2016 as per report of the Rwanda Environment Management Authorities (REMA 2014).

### 4.2. Impact of LULC change on soil erosion and sediment yield

The Sebeya watershed is more erosive with a R-factor value of 4339.5 MJ mm ha⁻¹ h⁻¹ yr⁻¹ greater than the global average 2000 MJ mm ha⁻¹ h⁻¹ yr⁻¹. The erosivity increase from the middle part to the eastern, while the lowland in the northwest recorded a low erosivity. Nevertheless, Kulimushi et al. (2021) stated that soil loss in

| Sub-watersheds | Area (ha) | MSY (t ha⁻¹ yr⁻¹) | TSY (%) | MSY (t ha⁻¹ yr⁻¹) | TSY (%) | MSY (t ha⁻¹ yr⁻¹) | TSY (%) | MSY (t ha⁻¹ yr⁻¹) | TSY (%) |
|---------------|-----------|-------------------|--------|-------------------|--------|-------------------|--------|-------------------|--------|
| Downstream    | 11362.94  | 1.08              | 26.07  | 3.64              | 14.04  | 11.99             | 34.31  | 9.23              | 19.83  |
| Karambo       | 8988.18   | 1.63              | 20.09  | 6.58              | 14.53  | 19.21             | 31.29  | 34.62             | 42.36  |
| Pfunda        | 4250.06   | 2.10              | 22.94  | 9.33              | 18.93  | 7.40              | 11.07  | 8.83              | 9.93   |
| Bihongoro     | 6072.38   | 1.43              | 13.09  | 8.63              | 14.42  | 7.15              | 8.82   | 9.39              | 8.69   |
| Upstream      | 5816.90   | 4.42              | 17.80  | 9.53              | 38.09  | 4.87              | 14.51  | 8.58              | 19.19  |

MSY: Mean sediment yield and TSY: Total sediment yield in percent.
tropical regions is not perfectly correlated with the R-factor. The Sebeya watershed is not an exception to the above statement because the current status of soil erosion shows high variation from the middle part toward western, northern, and southern inversely to the high variation of the erosivity from the central portion toward the eastern part of the watershed.

The slope was another natural factor to increase the severity of soil erosion in these highland regions, about 55% of the watershed has a slope value higher than 15° (33%). The result revealed an increase in LS value with the elevation from 2000 m and slope gradient from 15°, indicating shortness and steepness of the slope throughout the watershed. Cambisols are the predominant soil group in the Sebeya while per report of FAO (2001), Cambisols are found in eroding lands and hilly mountainous tropical regions. Therefore, the estimated soil erosion rate in the Sebeya mountainous landscape can also be attributed to its natural characteristics.

However, during the period under investigation, the other factors were kept constant and only the C-factor were changed in time due to the change in LULC, i.e. the spatio-temporal variation of the soil erosion was hence attributed to the C-factor which varied as the LULC changed in space and time (Balabathina et al. 2019; Özsahin and Eroğlu 2019). As a result, the soil erosion rate moved from 14.7 t ha\(^{-1}\) yr\(^{-1}\), 28.3 t ha\(^{-1}\) yr\(^{-1}\), and 34.0 t ha\(^{-1}\) yr\(^{-1}\) to 34.7 t ha\(^{-1}\) yr\(^{-1}\) from 1990, 2000, and 2010 to 2019, respectively.

The same ranges were recently noticed in comparative studies. For instance, the soil erosion rate in Rwanda was found around 39.2 t ha\(^{-1}\) yr\(^{-1}\) (Nambajimana et al. 2020), and 34.2 t ha\(^{-1}\) yr\(^{-1}\) (Fenta et al. 2020), 28 t ha\(^{-1}\) yr\(^{-1}\) and 37 t ha\(^{-1}\) yr\(^{-1}\) respectively in the Mukamira and Nyamyumba watersheds (Kabirigi et al. 2017), 38.4 t ha\(^{-1}\) yr\(^{-1}\) in the Satinskyi watershed (Byizigiro et al. 2020), and 41.5 t ha\(^{-1}\) yr\(^{-1}\) in the Buberuka highlands (Kagabo et al. 2013).

Furthermore, despite the geo-environmental settings being the same, soil erosion was lower in well-covered lands including grassland and forestland but higher in cropland. Though high rainfall, slope, and altitude observed from the eastern part of the watershed contrary to the middle and western parts with moderate rainfall and moderate altitude, however, these parts have a high proportion of cropland, and soil erosion in these regions was higher compared to that from the eastern parts with a high proportion of forestland and grassland.

Southgate and Whitaker (1992) stated, despite the high steep slopes and erosivity, the annual soil erosion is attenuated by effective and densely covered surface. Lal (1994) also added soil erosion is low in the forestland because the surface is covered and protected by vegetation and biomass. The latter increases the soil cohesion and reduce considerably the impact of the raindrops on soil (Prasannakumar et al. 2012; Esa et al. 2018). Furthermore, Trimble and Mende (1995) discovered a significant increase in intensive erosion rate as a response to extensive land cover clearing.

This demonstrates the influence of LULC to predict the past, actual, and future risk of soil erosion and inform the role plays by LULC to control and mitigate the severity of soil erosion. As stated by Alkharabsheh et al. (2013) the change in LULC cannot only be seen negatively, the author added the change in LULC can have a positive impact on soil erosion if the vegetation coverage increase in space and time.
This assumption was confirmed in this study, between 1990 and 2000, forestland lost about 38% of its area, as a consequence the soil erosion double i.e. from 14.7 t ha\(^{-1}\) yr\(^{-1}\) in 1990 to 28.3 t ha\(^{-1}\) yr\(^{-1}\) in 2000. Moreover, between 2000 and 2019 forestland lost about 32.8% and soil erosion increased by 6.4 t ha\(^{-1}\) yr\(^{-1}\). However, the increase of soil erosion was not proportional to the loss of forestland as compared to the 1990–2000 period. This was due to the high increase in grassland that attenuated the observed soil erosion. Grassland increased about 16.3% and has the lowest C-factor value (0.11). In addition, between 2010 and 2019 there was no change in soil erosion, the result showed an increase of about 0.7 t ha\(^{-1}\) yr\(^{-1}\), this is also attributed to the afforestation observed from the eastern part of the watershed in the 2019 land cover map.

The temporal change in soil erosion generated in cropland corresponded to the change observed in that land use category. About 7.4% of the total watershed area was occupied by cropland in 1990 and the generated soil erosion in cropland was about 44.48 t ha\(^{-1}\) yr\(^{-1}\) and contributed by 23.19%. However, after three decades i.e. in 2019, the cropland accounted for 42% of the watershed area and the soil erosion in cropland increased to 67.82 t ha\(^{-1}\) yr\(^{-1}\) and shared 82.38% of the total soil erosion. Furthermore, in 1990 only 5.35% of the watershed area was exposed to soil erosion greater than 30 t ha\(^{-1}\) yr\(^{-1}\) (high and very high severities) but actually, about 32.18% of the total watershed area is exposed to high and very high soil erosion and they are responsible for 84.76% of the total soil loss in 2019.

Several researchers have found similar results, they predicted a high influence of LULC change on soil erosion (Kavian et al. 2017; Gashaw et al. 2019; Özşahin and Eroğlu 2019; Phinzi and Ngetar 2019). The detrimental impact of LULC change in developing countries was noticed earlier by McCool and Renard (1990). In addition, Speth (1994) agreed with this assumption, the author reported that almost 80% of the world’s agricultural land is exposed to severe soil erosion.

Furthermore, Borrelli et al. (2017) strongly approved the findings, the study revealed that cropland is likely to produce a soil erosion rate up to 77 times higher than that observed in natural forests. The same views are shared in Tsegaye (2019) and Aneseyee et al. (2020), the authors observed an increase in soil erosion as the cultivation area was continuously increasing in space and time. W4GR (2018), through an integrated water resource management program in the Sebeya watershed also supports these findings, he reported about 18 000 ha (50%) of the entire watershed are exposed to extreme erosion risk mainly accelerated by unsustainable farming practices and inadequate cattle drenching among other. NISR (2018) also reported, the temporal increase in cultivated land at the expense of healthy forest contributed considerably to accelerating runoff and soil loss in Rwanda.

Apart from the influence of the change in land use/cover on soil erosion, as discussed above, sediment yield reflects the soil erosion. The sediment yield was also aggravated by the stream slope which resulted in the high range of SDR value but the temporal change in the mean sediment yield demonstrated the impact of LULC change. As a result, the mean sediment yield moved from 4.6 t ha\(^{-1}\) yr\(^{-1}\), 7.26 t ha\(^{-1}\) yr\(^{-1}\), and 9.8 t ha\(^{-1}\) yr\(^{-1}\), to 13 t ha\(^{-1}\) yr\(^{-1}\) from 1990, 2000, and 2010 to 2019 respectively. This range is similar to that recently found in Ethiopia. The mean sediment yield ranging from 6.7 to 9.4 t ha\(^{-1}\) yr\(^{-1}\) was reported by (Kidane et al. 2019), and from14.8 to 22.1 t ha\(^{-1}\) yr\(^{-1}\) by Gashaw et al.
In addition, Azanga et al. (2016) reported, the LULC change was the key driving factor to increasing the sediment yield in the Lake Tanganyika basin. The spatio-temporal variation of cropland compromised the watershed integrity, and expose this to severe erosion which coupled with the nature of topography enlarge volume of sediment deposition into reservoirs and thereby making water unsuitable for consumption and irrigation due to possible eutrophication from ingress of fertilizer from agricultural land (Christian et al. 2012). Coxhead and Shively (2005) reported, in a tropical area, the mean sediment yield is sensitive to human disturbance on natural resources, the author showed that sediment yield double increases if cropped area doubles in space and time. The current contribution of the sub-watersheds to the total sediment yield revealed that the highest loading into the mainstream Sebeya River follows the sub-watersheds order: Karambo, Downstream, Upstream, Pfunda, and Bihongoro responsible for 42.3%, 19.8%, 19.1%, 9.9%, and 8.6% to the total sediment yield.

Uwacu and Olalekan (2019) assessed water quality at a sub-watershed scale and reported a similar result. The author reported, the turbidity values in Karambo and Sebeya upstream sub-watersheds were respectively 2330 and 1570 NTU compare to the standard of about 5 NTU. The total suspended solids (TSS) of 1700 and 1360 mg/l respectively as compared to the standard of less than 30 mg/l. Total nitrogen (TN) of 39.2 and 27.6 mg/l respectively, higher than the standard of less than 3 mg/l. Moreover, the chemical oxygen demand (COD) of 105 and 78.5 mg/l against the limit of <50 mg/l. The entire sub-watersheds were found higher than the international standards for physical and chemical parameters of water (WHO 2004; RSB 2008; Uwacu and Olalekan 2019).

As a consequence of rivers sedimentation in the Sebeya Rivers, it damages the hydropower infrastructure and regularly interrupts services during the period of heavy rainfall as stated (W4GR2018). In addition, the river’s sedimentation increase the water overflowing and downstream flooding, increase drinking water treatment cost and offsite lake contamination (Pimentel and Burgess 2013). Therefore, proper land use planning and the adoption of adequate conservation measures are important to mitigate the future consequences of soil erosion and sedimentation of rivers.

5. Conclusions

The present study evaluated soil erosion and sediment yield spatio-temporal pattern from 1990 to 2019 due to LULC change using GIS software, multi-temporal land cover maps derived from RCMRD, and Landsat data in the RUSLE model. The study noticed considerable change in LULC classes. Forestland and water bodies decreased whereas cropland, grassland, and settlement have increased. During the period 1990–2019, forestland lost in total 25376.3 ha equivalent to an annual decrease of about 845.9 ha/year (2.6%). While grasslands increased in total about 11292.4 ha approximately to an annual increase of 376.4 ha/year (23.6%), cropland increased about 12466.5 ha equivalent to the annual rate of about 415.5 ha/year (14.5%) and settlement increased 1617 ha corresponding to 53.9 ha/year (81.6%) of annual increasing rate.

The result indicated a temporal variation in soil erosion due to LULC change. The mean soil erosion for 1990, 2000, 2010, and 2019 were 14.7 t ha⁻¹ yr⁻¹, 28.3 t
ha$^{-1}$yr$^{-1}$, 34 t ha$^{-1}$yr$^{-1}$, and 34.7 t ha$^{-1}$yr$^{-1}$ respectively whereas the mean sediment yield moved from 4.68 t ha$^{-1}$yr$^{-1}$, 7.26 t ha$^{-1}$yr$^{-1}$, and 9.8 t ha$^{-1}$yr$^{-1}$ to 13 t ha$^{-1}$yr$^{-1}$ respectively. Cropland was an exception to the other LULC classes because it has generated soil erosion in very high severity $>$50 t ha$^{-1}$yr$^{-1}$. Cropland was therefore considered as highly susceptible land use. Furthermore, the temporal comparison of the soil erosion in different LULC classes indicated that forestland and cropland conversely to grassland and settlement have experienced an increase in soil erosion rate but the rate was considerable in cropland. In addition, the spatial distribution of soil erosion severity classes suggested an increasing trend of high and very high whereas very low, low, and moderate showed a downward trend.

The study suggests that strong measures should be implemented in the watershed to mitigate the future consequences of soil erosion. The study encouraged the afforestation observed in the 2019 land cover map that considerably mitigated the soil erosion which increased by only (+0.7 t ha$^{-1}$yr$^{-1}$) between 2010 and 2019. This effort is highly encouraged throughout the watershed mostly in the central part towards western parts vulnerable to high and very high erosion risk. Moreover, proper land-use planning is at most important to slow the high deforestation rate. Adopting terracing, agroforestry, and intelligent agriculture, and constructing check dams at the sub-watershed mouth are highly encouraged. Sub-watersheds should also be prioritized by focusing on critical areas; future research should focus on detecting and mapping the potential regions for afforestation of the high-risk areas.

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Disclosure statement

The authors reported no potential conflict of interest.

Data availability statement

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

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Appendix

Table A1. K-factor values as a function of organic matter content and the soil textural class adapted from Stone and Hilborn (2012).

| Texture class   | Average | Less than 2% | More than 2% |
|-----------------|---------|--------------|--------------|
| Sand            | 0.02    | 0.03         | 0.01         |
| Loamy sand      | 0.04    | 0.05         | 0.04         |
| Sandy loam      | 0.13    | 0.14         | 0.12         |
| Loam            | 0.3     | 0.34         | 0.26         |
| Silt loam       | 0.38    | 0.41         | 0.37         |
| Silt            | 0.6     | 0.43         | 0.6          |
| Sandy clay loam | 0.2     |              | 0.2          |
| Clay loam       | 0.3     | 0.33         | 0.28         |
| Silt clay loam  | 0.32    | 0.35         | 0.3          |
| Sandy clay      | 0.14    | 0.1          | 0.14         |
| Silt clay       | 0.26    | 0.27         | 0.26         |

Table A2. Confusion matrix result of LULC 2019.

| C1  | C2  | C3  | C4  | C5  | Total | U.A     | CE     | OA    | Kappa |
|-----|-----|-----|-----|-----|-------|---------|-------|-------|-------|
| (C1) Forestland | 30  | 2   | 0   | 0   | 32    | 93.75% | 6.3%  | –     | –     |
| (C2) Grassland   | 9   | 30  | 2   | 0   | 41    | 73.17% | 26.8% | –     | –     |
| (C3) Cropland    | 1   | 8   | 33  | 5   | 47    | 70.21% | 29.8% | –     | –     |
| (C4) Settlement  | 0   | 0   | 5   | 35  | 40    | 87.50% | 12.5% | –     | –     |
| (C5) Waterbody   | 0   | 0   | 0   | 15  | 15    | 100.00%| 0.0%  | –     | –     |
| Total            | 40  | 40  | 40  | 40  | 175   | –       | –     | –     | 81.7% |

OA: Overall accuracy, PA: Producer accuracy, U.A: User accuracy, CE: Commission error, OE: Omission Error.

Figure A1. Average annual rainfall during 1990–2019 periods.
Figure A2. Average monthly rainfall during 1990–2019 periods.