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Modelling and mapping erosion in smallholder agro-ecosystems, Tanzania

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Abstract

The West Usambara Highlands in north-eastern Tanzania have many smallholder agro-ecosystems with unknown composition, management, and vulnerability to erosion. Their specific locations and spatial extent are difficult to trace by satellite images or remote sensing imagery alone. To address these limitations, we combined ground soil surveys, geographic information system, and erosion modelling to (a) locate and map smallholder agro-ecosystems, (b) determine their biophysical characteristics, and (c) model their soil losses. Land resource information was collected from 301 random 0.1-ha plots sampled from a total area of 200 km^2. Annual soil losses were estimated using the universal soil loss equation. The study located six dominant agro-ecosystems with the following spatial extent: maize-bean (24.9%), maize-bean-agroforestry (31.2%), maize-bean-agroforestry-high value trees (18.9%), tree farms (7.0%), forests (15.6%), and grazing lands (2.3%). Agroforestry and other tree-based agro-ecosystems dominate the area due to historical land use change and later institutional interventions. This study finds combined use of soil surveys, geographic information system, and modelling to be reliable in locating, mapping, and assessing soil losses in smallholder agro-ecosystems. The agro-ecosystems differ significantly ($p < 0.05$) in slope, vegetation cover, soil conditions, and soil losses. Soil loss in the maize-bean agro-ecosystem (28.3 t ha^{-1} yr^{-1}) was 18 times higher compared with natural forests (1.57 t ha^{-1} yr^{-1}) due to lower soil cover and inefficient conservation and cultivation practices. Our results show that adoption of soil conservation measures and improved vegetation cover technologies across the agro-ecosystems reduces soil losses by 37% and increases organic carbon levels by 16%.

KEYWORDS

agro-ecosystems, biophysical factors, soil erosion, Tanzania, West Usambara Highlands

1 | INTRODUCTION

Agro-ecosystems are communities of plants and animals interacting with their physical and chemical environments to produce food, fibre, fuel, and other products for human consumption and processing (Altieri, 2002). Though agro-ecosystems are important for food and nutritional security in Sub-Saharan Africa, their capacity to produce food is currently undermined by many factors, including soil degradation (Tamene & Le, 2015). Soil erosion is the most visible form of degradation in smallholder agro-ecosystems and has been widely linked to inefficient land management and resource utilization by farmers (Adgo, Teshome, & Mati, 2013). However, for East African highlands, the extent and magnitude of soil erosion in specific smallholder agro-ecosystems has not been adequately studied. Furthermore, due the variation in agro-ecosystems in the region, erosion is location specific and cannot be generalized (Grogan, Birch-Thomsen, & Lyimo, 2013).

The West Usambara Highlands (WUH) in Tanzania are among the most eroded mountains in East Africa (Nyanga, Kessler, & Tenge, 2016). A recent study in the area linked current erosion to land management practices by smallholder farmers (Winowiecki et al., 2016). Past studies in the area indicate that farming in the WUH is mainly based on smallholder farming in the following agro-ecosystems: maize-bean, traditional agroforestry and maize-bean-agroforestry (Johansson, 2001), maize-bean and coffee-banana-high value trees (Tenge, De Graaff, & Hella, 2004), tree farms (Wickama & Nyanga, 2009), and natural forests and agro-pastoralist systems (Mbogoni, 2010). Though most soil erosion reports covering the WUH (e.g. Vigiak, Okoba, Sterk, & Groenenberg, 2005; Vrieling, Sterk, & Vigiak, 2006) conclude that soil erosion in the area is a land management problem, most have not considered the complex composition of existing agro-ecosystems.

Though information about spatial extent and complexity of agro-ecosystems is very useful for planning soil erosion interventions, this knowledge is scant in places like the WUH. Application of remote sensing (RS) techniques could potentially assist in classifying and mapping agro-ecosystems. However, the generally high vegetation cover, the similarity in vegetation types, and the scattered distribution of the agro-ecosystems make it very difficult to detect and map them adequately. As a result, most RS applications used in East Africa for detecting erosion in different agro-ecosystem have only discriminated between natural forests, agricultural crop land, and grazing land (Bezuayehu & Sterk, 2008; Biazin & Sterk, 2013). This also counts for two other recent studies in the WUH. Vrieling et al. (2006) used the normalized difference vegetation index and a slope map to predict soil erosion risk. Wickama, Masselink, and Sterk (2015) used RS and object-based image analysis to detect soil and water conservation (SWC) measures at the landscape scale and model soil erosion rates. As both studies did not discriminate between different agro-ecosystems, there is a knowledge gap concerning the location and magnitude of soil losses in the different agro-ecosystems of the area. In order to fill this knowledge gap, this study applied ground-based soil surveys, geographic information system (GIS) tools, and soil erosion modelling to (a) locate and map smallholder agro-ecosystems, (b) determine bio-physical characteristics of dominant agro-ecosystems, and (c) assess soil losses in these agro-ecosystems.

2 | MATERIALS AND METHODS

2.1 | Description of study area

This study was conducted in the Lushoto District in Tanzania, within latitudes 4°22′ to 5°08′ south and longitudes 38°5′ to 38°38′ east. The district is 3,500 km^2 in size with a population density of 134 people per km^2 (Wickama, Okoba, & Sterk, 2014). Its topography is mountainous, with an altitude that varies from 600 to 2,300 m above sea level (asl; Wickama et al., 2015). Its major soils are Acrisols, Luvisols, and Lixisols on the hills and midslopes, whereas valley bottoms are dominated by Fluvisols with pockets of Gleysols (Wickama et al., 2014). The district has two rainfall seasons. The long rains ("masika") start mid-March to end of June, while short rains ("vuli") start mid-October ending late December. Annual rainfall varies from 800 to 1,300 mm depending on the agro-ecological zone (Mascarenhas, 2000). Agriculture in the district is low-input subsistence farming.

The study considered six agro-ecosystems commonly found in the WUH (Figure 1), namely, (a) maize-bean, (b) maize-bean-agroforestry, (c) maize-bean-agroforestry-high value trees, (d) tree farms, (e) natural forests, and (f) grazing lands. Natural forests were also used as a control group for comparison of soil conditions and erosion levels with the other agro-ecosystems.

Details about these six agro-ecosystems can be found in Data S1. Field data were collected from two villages (Soni and Sunga) covering an area of 100 km^2 each. These villages were selected because all six agro-ecosystems are found in each village, their soil characteristics are typical of the WUH, and both have good accessibility. Furthermore, they represent the typical ecological and smallholder land use challenges commonly found in the East African region as has been reported from Kenya (Vigiak et al., 2005), Ethiopia (Tamene, Adimassu, Ellison, et al., 2017), and Rwanda (Kagabo, Stroosnijder, Visser, & Moore, 2013). For this reason, results obtained in these two sites were considered representative and up-scalable. Soni village is located 12 km east of Lushoto town, whereas Sunga is 50 km north-west of Lushoto (Figure 2). For Soni, data were collected from a square area (4°48′–4°53′ south and 38°20′–38°25′ east) surrounding the village. Similarly, for Sunga, the data were collected from the square defined by 4°29′–4°34′ south and 38°12′–38°17′ east.

To locate and map agro-ecosystems, two ground-based soil surveys were conducted covering 301 randomly selected plots: 160 in Soni and 141 in Sunga. These plots were established following the approach described by Vågen et al. (2004). First, 16 primary sampling units were randomly selected from each 100 km^2 study site (Figure 2). Then, 10 plots of 0.1 ha each were randomly sampled from each of these primary sampling units. In each of these plots, measurement of altitude (by Global Positioning System unit), surface slopes (inclinometer), vegetation types and cover (visual assessment), major crops (visual assessment), and dominant erosion features (visual) were taken following methods modified from Watson (1985) in Scotland (Table 1). Maize and bean yields were estimated from farmers own data. Each random plot was classified as belonging to any of the six agro-ecosystems if at least 50% of observed features on the plot belonged to that agro-ecosystem.

Soil samples were collected from the 301 plots at two depths: 0–20 cm for top soils and 30–50 cm for subsoils. These two depths were used because they represent the active root zone for most tropical crops. Samples were composite, meaning that four topsoil samples from different locations within a random plot were sampled and mixed as one. The same was done for the subsoil samples. Collected soils were air-dried and sieved (2-mm sieve). A small portion (10%) of the soil samples was analysed for particle size distribution, soil reaction (pH), soil organic carbon, total nitrogen, exchangeable bases, and available phosphorus. Respective analytical methods used for each parameter are described in Page (1982). Analytical results of the 10% portion were then used for calibration and analysis of the remaining 90% through near-infrared spectroscopy (Brown, Shepherd, Walsh,

FIGURE 1 Selected agro-ecosystems in the West Usambara Highlands, Tanzania [Colour figure can be viewed at wileyonlinelibrary.com]

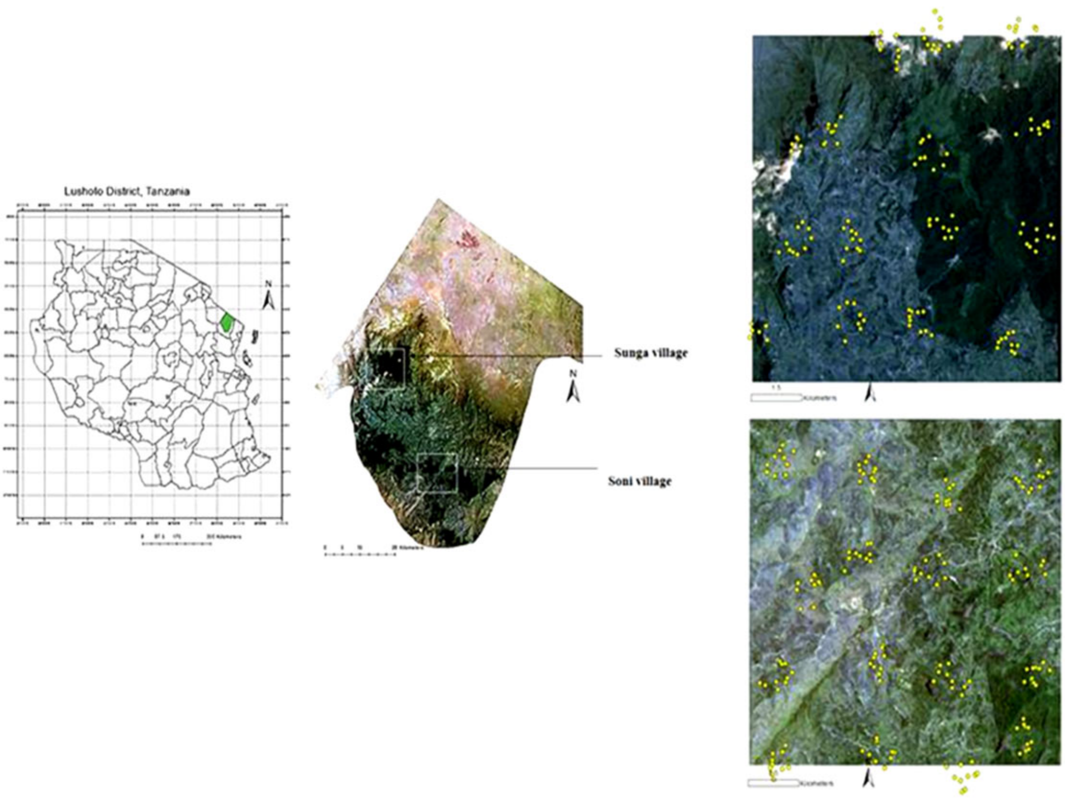

FIGURE 2 Location of Lushoto district, the two study villages, and sampled locations (yellow dots) in each of the 16 primary sampling units [Colour figure can be viewed at wileyonlinelibrary.com]

TABLE 1 Criteria of scoring soil erosion features in Lushoto District, Tanzania

Visible erosion features	Score
None	1
Sheet erosion	2
Rills	3
Sheet and rills	4
Gullies	5

Source: Adapted from Watson (1985).

Dewayne Mays, & Reinsch, 2006). Near-infrared analysis was used for the same soil properties as for standard laboratory analysis, except for soil texture.

2.2 | Modelling of soil erosion

Soil losses were estimated by the universal soil loss equation (USLE) model (Wischmeier & Smith, 1978). Despite known limitations in representing erosion and hydrologic processes (Morgan, 2005), this model is still preferred for soil erosion modelling (Renard, Foster, Weesies, & Porter, 1991). The USLE has also been widely used in the East African highlands (Angima, Stott, O'Neill, Ong, & Weesies, 2003; Mati et al., 2000), thus enabling comparison of our results to those obtained elsewhere in the region. It is defined as

$$A = R \times K \times L \times S \times C \times P, \tag{1}$$

where A is annual erosion (t ha^{-1} year^{-1}), R is rainfall and run-off factor (MJ mm ha^{-1} hr^{-1} year^{-1}), K is soil erodibility factor (t ha^{-1} per unit of R), L is slope length factor (−), S is slope steepness factor (−), C is cover and management factor (−), and P is support practice factor (−). Rainfall data were collected from automatic weather stations installed in each village for 30 months (January 2010 to June 2012). Annual mean rainfall during this period was 691.3 and 700.4 mm for Soni and Sunga respectively. R factors were then calculated as described by Moore (1979) in which

$$KE = 11.46P_a - 2,226, \tag{2}$$

where KE represents kinetic energy (J m^{-2}) of the rain and P_a represents the mean annual rainfall (mm). Then the R factor (MJ mm ha^{-1} hr^{-1} year^{-1}) was calculated as

$$R = 0.029KE - 26.0. \tag{3}$$

The K factor (t ha^{-1} per unit of R) was calculated from the equation proposed by Lal and Elliot (1994):

$$K = 2.8{\times}10^{-7}M^{1.14}(12{-}a) + 4.3{\times}10^{-3}(b{-}2) + 3.3{\times}10^{-3}(c{-}3), \tag{4}$$

where a is the percentage soil organic matter, b is a soil structure class (1 to 4), and c is a soil profile permeability class (1 to 6). M is the particle size parameter, which is calculated using the mass fractions of clay, silt, and fine sand:

$$M = (\%silt + \%fine\ sand) \times (100{-}\%clay). \tag{5}$$

For all locations, we used the soil structure class with a value of 2 (meaning moderate) and the soil permeability class with a value of 3 (meaning moderately slow). K factors were calculated from sampled soils. C-factor values were derived from vegetation cover at each random plot. Values of vegetation cover (cov in %) at each random plot were converted into C-factor values by using an equation tested in China (Ma, Xue, Ma, & Wang, 2003). Choice of this equation was based on its convenience of collecting vegetation cover data in the field during soil survey assessments instead of soliciting satellite imagery for each location:

$$C = 0.6508{-}0.343 log cov. \tag{6}$$

This equation works best for vegetation cover not exceeding 78.3%. For those cases where vegetation cover exceeded this limit, a recommended C-factor value was adopted from the USLE manual (Wischmeier & Smith, 1978). In our study, collection of vegetation cover data was conducted during off-rain-season periods to enable logistical access and to accurately estimate erosion risk of individual locations without influence of seasonal crop cover.

The L factor was estimated using Wischmeier and Smith (1978):

$$L = \left(\frac{\lambda}{22.13}\right)^{0.5}, \tag{7}$$

where λ is slope length (m). For this study, λ was assumed to be 12.23 m after making adjustments for slope (Walsh, Shepherd, Awiti, & Vagen, 2006). An exponent value of 0.5 was chosen because of steep slopes (>30%) in both villages.

S-factor values were calculated using the following equation proposed by Ma et al. (2003):

$$S = 0.8252 \times 8.5319 sin\theta, \tag{8}$$

where θ is slope angle (°). This S-factor equation was chosen for its strength of reducing exaggeration of LS values in steep landscapes (Ma et al., 2003).

The P-factor represents soil loss in the absence of SWC measures. Specifically, plots without bench terraces or grass strips were given a P value of 1. Those with bench terraces or grass strips were given P values ranging from 0.10 to 0.18 (Mati et al., 2000). P-factor values of 0.10 were assigned to random plots with more than 40 bench terraces or grass strips.

All USLE values (R, K, L, S, C, and P) were computed in Microsoft EXCEL. These losses were then summarized into soil losses per agro-ecosystem. Spatial distribution of the agro-ecosystems and their corresponding soil losses were mapped using ArcGIS 10.1. Severity of soil losses was classified using a criterion proposed for its categorization by Stone and Hilborn (2000).

2.3 | Statistical analyses

Statistical analyses were conducted using the Microsoft EXCEL Data Analysis toolbox and R software. Student t tests and one-way analysis of variance were used to compare means of samples (slope, vegetation cover, and soil properties) originating from the six agro-ecosystems. The means of the non-parametric erosion classes and the numbers of SWC measures in the random plots were compared using the Wilcoxon rank sum test in *R*. All tests were done using a significance level (α) of 0.05. Correlation analysis between modelled annual soil losses and several bio-physical properties was carried out using the Pearson correlation coefficient for parametric data and the Spearman's rank correlation coefficient for non-parametric data.

3 | RESULTS

3.1 | Location and spatial extent of agro-ecosystems

Locations of the studied agro-ecosystems are presented in Figure 3. Their respective spatial extent, altitude, and estimates of maize and bean yields are given in Table 2. Average altitude for agro-ecosystems located in Sunga is higher (1,781.6 m asl) than in Soni (1,408.6 m asl), whereas average temperature in Sunga (20°C) is lower than in Soni (22°C). Annual rainfall for agro-ecosystems in Sunga is slightly higher (700.4 mm year^{-1}) compared with Soni (691.3 mm year^{-1}; Wickama et al., 2015).

In Soni village, the maize-bean agro-ecosystem occupied central locations in close proximity to maize-bean-agroforestry fields. Tree farms and natural forests were on eastern fringes of Soni village. In both areas tree farms and natural forests occupied highest altitudes. In Sunga village, unlike Soni, the maize-bean agro-ecosystem occupied peripheral areas of the village and was more fragmented (Figure 2) whereas natural forests and tree farms in Sunga village occupied eastern and central locations.

In both villages, maize-bean-agroforestry-high value trees fields were located closer to tree farms and natural forests than were maize-bean and maize-bean-agroforestry fields. In Sunga, natural forests (Shagayu Forests) occupied approximately 20% of the village area, whereas in Soni village, the forest (Ndelemai Forest) occupied 11% of the village. Tree farms were twice as large in Sunga (Table 2) compared with those in Soni village, due to the presence of a commercial plantation (Shagayu Plantation) and a saw mill in Sunga. Grazing lands had no specific locations but were small in size and often in small pockets of open areas between major landscape units.

Overall, different types of maize-bean-agroforestry fields dominated both villages (>30%). In Soni village, this most basic agro-forestry system occupied about 42% of the area, whereas its more advanced form (the maize-bean-agroforestry-high value trees agro-ecosystem) occupied only 9.4%. In Sunga, the maize-bean-agroforestry fields cover 19.2% (Table 2), whereas the maize-bean-agroforestry-high value trees fields are more widespread (29.8%) than in the Soni area. Grazing lands covered less than 4% in both villages, thus making free grazing difficult.

3.2 | Bio-physical characteristics of the dominant agro-ecosystems

Bio-physical characteristics of the six major agro-ecosystems are shown in Table 3. Except for Grazing lands, all agro-ecosystems are on steep (>35%) slopes, with maize-bean and maize-bean-agroforestry fields being located on the steepest slopes (>40%). Vegetation cover was highest in natural forests (average of 94.7%) and lowest in the maize-bean agro-ecosystem (23.8%). Vegetation cover was

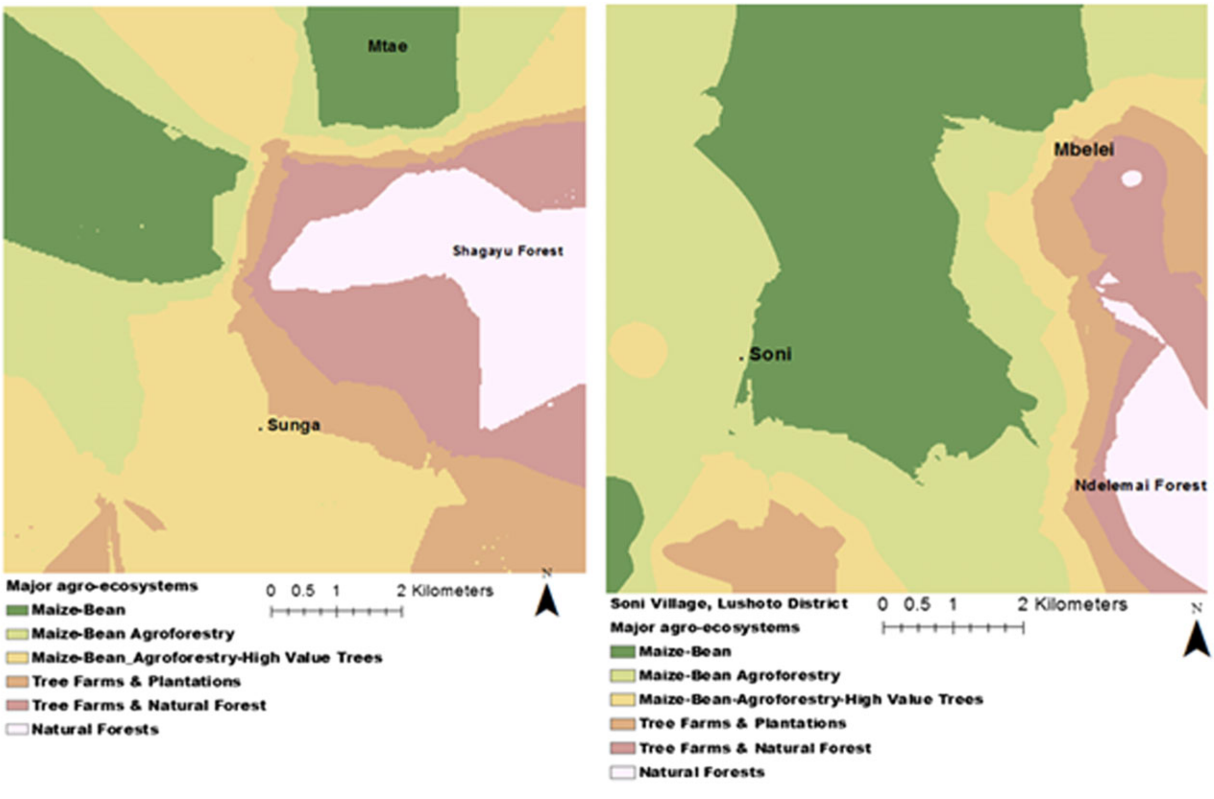

FIGURE 3 Location of major agro-ecosystems in Sunga and Soni villages (Lushoto District Tanzania) [Colour figure can be viewed at wileyonlinelibrary.com]

TABLE 2 Spatial extent, altitude, and estimates of maize and bean yields of major agro-ecosystems in Soni and Sunga villages (Lushoto District, Tanzania)

	Soni					Sunga				
	Plots	Extent	Altitude	Yield		Plots	Extent	Altitude	Yield	
Agro-ecosystem	*n*	%	m asl	Maize kg ha^{-1}	Bean kg ha^{-1}	*n*	%	m asl	Maize kg ha^{-1}	Bean kg ha^{-1}
Ma-Be	51	31.9	1,313	596	455	24	17.0	1,615	770	728
Ma-Be-Af	67	41.9	1,377	648	468	27	19.2	1,632	887	792
Ma-Be-Af-HVT	15	9.4	1,486	642	414	42	29.8	1,822	860	720
Tree farms	7	4.4	1,535	218	120	14	9.9	1,963	243	196
Natural forests	18	11.3	1,693	-	-	29	20.6	1,920	-	-
Grazing lands	2	1.3	1,303	-	-	5	3.6	1,741	-	-

Note. Ma-Be: maize-bean; Ma-Be-Af: maize-bean-agroforestry; Ma-Be-Af-HVT: maize-bean-agroforestry-high value trees.

TABLE 3 Bio-physical characteristics of the major agro-ecosystems of Lushoto District, Tanzania

Agro-ecosystem	Plots	Slope	Erosion class	SWC measures				Vegetation cover
						Plots without SWC		
	n	%	Average Score	Average Number	Max Number	*n*	%	%
Ma-Be	75	40.1a	2.33a	5.93a	38	52	69.3	23.8a
Ma-Be-Af	94	40.3a	2.10ab	8.71b	47	49	52.7	34.2b
Ma-Be-Af-HVT	57	35.8b	2.02b	10.53BC	34	26	45.6	35.8b
Tree farms	21	39.0ab	1.48c	2.86a	34	17	81.0	60.7c
Natural forests	47	37.1b	1.31c	0d	0	47	100	94.7d
Grazing lands	7	10.8c	2.14abd	2.57ab	10	5	71.4	48.7BC

Note. Ma-Be: maize-bean; Ma-Be-Af: maize-bean-agroforestry; Ma-Be-Af-HVT: maize-bean-agroforestry-high value trees. Values in column followed by the same letter are not significantly (α = 0.05) different.

significantly higher in the maize-bean-agroforestry and maize-bean-agroforestry-high value trees systems than in the maize-bean systems. Erosion was most serious in the maize-bean, grazing lands, and maize-bean-agroforestry agro-ecosystems. In the maize-bean and maize-bean-agroforestry agro-ecosystems, this can be explained by the relatively steeper slopes and lower vegetation cover. In the case of grazing lands, livestock trampling causes topsoil compaction, which reduces the infiltration capacity of the soil, resulting in high erosion rates. The lowest erosion scores were obtained in tree farms and natural forests, where high vegetation cover protects soils from erosion. SWC measures were most abundant in the maize-bean-agroforestry-high value trees agro-ecosystems, followed by maize-bean-agroforestry and maize-bean.

Main soil properties in the six agro-ecosystems are given in Table 4. Except for clay content, which is in most cases much higher in subsoils (B horizon) than in top soils, most soil properties are nearly equal within the first 50 cm of the soil. The only exception is for natural forests, where top soils had substantially higher organic matter

TABLE 4 Soil properties in the major agro-ecosystems of Lushoto District, Tanzania

Agro-ecosystem	Soil depth (cm)	Clay (%)	Ex. Ca (me/100 g)	Ex. K (me/100 g)	Ex. Mg (me/100 g)	Av. P (mg/kg)	SOC (%)	pH(-)	Total N. (%)
Ma-Be	0–20	38.5	8.66	0.74	0.49	2.40	2.04	5.9	0.14
	30–50	43.6	8.28	0.72	0.47	2.40	2.01	5.9	0.14
Ma-Be-Af	0–20	37.3	7.25	0.77	0.41	2.43	2.10	5.9	0.14
	30–50	44.0	7.05	0.80	0.40	2.46	2.16	5.9	0.15
Ma-Be-Af-HVT	0–20	29.7	5.89	1.01	0.36	2.53	2.37	5.7	0.15
	30–50	41.1	5.68	1.05	0.35	2.54	2.38	5.7	0.15
Tree farms	0–20	23.3	5.76	1.04	0.33	2.60	2.77	5.6	0.17
	30–50	32.7	6.25	1.10	0.37	2.55	2.69	5.6	0.17
Natural forests	0–20	10.6	4.49	1.74	0.20	3.19	5.37	5.1	0.49
	30–50	51.2	5.17	1.36	0.23	2.87	4.60	5.2	0.37
Grazing lands	0–20	36.0	4.85	1.05	0.24	2.71	3.01	5.6	0.23
	30–50	36.1	4.78	0.90	0.25	2.51	2.40	5.6	0.17

Note. Av. P: available phosphorus; SOC: soil organic carbon. Ma-Be: maize-bean; Ma-Be-Af: maize-bean-agroforestry; Ma-Be-Af-HVT: maize-bean-agroforestry-high value trees.

levels (soil organic carbon) than the subsoil, resulting in higher N, P, and K levels. When considering the main agro-ecosystems, there were only marginal differences between maize-bean, maize-bean-agroforestry, and maize-bean-agroforestry-high value trees systems.

3.3 | Soil erosion in agro-ecosystems

The USLE-related soil erosion factors as well as the calculated annual soil losses per hectare for each agro-ecosystem are presented in Table 5. The different number of random plots in the two villages for the same agro-ecosystem (Tables 2 and 3) caused the average *R* factor being different for each agro-ecosystem. *C*-factor values were high in maize-bean agro-ecosystem and low in natural forests. *LS*-factor values were highest in maize-bean-agroforestry and maize-bean agro-ecosystems, and low in grazing lands, which are located on more moderate slopes. The lowest *P*-factor values were observed in maize-bean-agroforestry-high value trees due to the higher number of SWC measures observed in this agro-ecosystem. The low presence of such measures in maize-bean and grazing lands resulted in relatively high *P* values, whereas the highest values were obtained for tree farms (P = 0.84) and natural forests (P = 1).

Annual calculated soil losses ranged from a minimum of 0.04 t ha^{-1} yr^{-1} in grazing lands to a maximum of 112.8 t ha^{-1} yr^{-1} in maize-bean agro-ecosystem and occurred in three groups of magnitude. The highest soil losses were obtained in the maize-bean agro-ecosystem (28.3 t ha^{-1} yr^{-1}). The next group, with soil losses varying from 16.4 to 17.9 t ha^{-1} yr^{-1}, comprises maize-bean-agroforestry, maize-bean-agroforestry-high value trees, and tree farms agro-ecosystems. Smallest erosion losses were predicted for natural forests and grazing lands, which had soil losses of less than 6.6 t ha^{-1} yr^{-1}. Most modelled values followed the erosion scores that were observed in the random plots (Table 3). However, grazing lands received an average erosion score similar to maize-bean-agroforestry, whereas modelled erosion for Grazing lands was much lower. This discrepancy is probably because the model does not include effects of trampling on the soil and resulting reduction in infiltration capacity. In the case of tree farms, its moderately high erosion values were caused by the relatively high values of *LS* in that agro-ecosystem.

Results from correlation analysis (Figure 4) shows that for all agro-ecosystems, soil losses were negatively correlated to the percentage of vegetation cover and the number of SWC measures present in a random plot. Except for grazing lands, which had low values for slopes and *LS* factor (Tables 3 and 5), the slope showed only a moderate positive correlation with annual soil loss.

In both villages, severe erosion was observed in hamlets where maize-bean agro-ecosystems dominated (Figure 5). However, soil losses were small in locations with natural forests, tree farms, grazing lands, and maize-bean-agroforestry-high value trees. Smaller soil losses were also observed in areas with high concentrations of SWC measures.

4 | DISCUSSION

Predominance of agro-forestry-related systems in our study villages is related to afforestation efforts from the early 1980s to 2000 through a project called Soil Erosion Control and Afforestation Project (SECAP). Through SECAP 10 million trees were planted and 79,000 ha of farmland were put under agro-forestry (Johansson, 2001). The

TABLE 5 Variation of USLE-soil erosion factors and predicted soil losses for the major agro-ecosystems in Lushoto district, Tanzania

Agro-ecosystem	Property	R factor (MJ mm ha^{-1} hr^{-1} $year^{-1}$)	*C* factor	*LS* factor	*P* factor	*K* factor (t^{-1} ha^{-1} per unit of R)	Soil loss (t^{-1} ha^{-1} $year^{-1}$)
Ma-Be	Mean	2,499.8b	0.2210a	2.0034ab	0.74BC	0.0332BC	28.3a
	Min	2,369.62	0.1346	0.73	0.11	0.0143	1.2
	Max	2,776.32	0.3448	3.01	1	0.0738	112.8
	SD	189.71	0.0400	0.57	0.4	0.0125	25.7
Ma-Be-Af	Mean	2,486.4b	0.1673b	2.0185a	0.59cd	0.0355b	17.8b
	Min	2,369.62	0.0637	0.73	0.1	0.0019	0.21
	Max	2,776.32	0.2844	3.01	1	0.0738	112.8
	SD	184.02	0.0409	0.53	0.43	0.01449	20.3
Ma-Be-Af-HVT	Mean	2,669.3a	0.1679b	1.8055c	0.53d	0.0400a	17.9b
	Min	2,369.62	0.0095	1.01	0.11	0.0019	0.2
	Max	2,776.32	0.2573	3.11	1.00	0.0594	99.4
	SD	179.09	0.0587	0.49	0.43	0.0135	21.1
Tree farms and plantations	Mean	2,640.8a	0.0921 c	1.95abc	0.84b	0.0426a	16.4bc
	Min	2,369.62	0.0192	0.82	0.11	0.01841	0.5
	Max	2,776.32	0.2492	3.21	1.00	0.05380	78.3
	SD	191.72	0.0758	0.55	0.33	0.00888	17.9
Natural forests	Mean	2,620.6a	0.0101d	1.87BC	1.00a	0.0309c	1.6d
	Min	2,369.62	0.0018	1.01	1.0	0.0206	0.2
	Max	2,776.32	0.0314	2.71	1.0	0.0493	7.6
	SD	197.7	0.0074	0.43	0.0	0.0081	1.42
Grazing lands	Mean	2,660.1a	0.1561b	0.56d	0.7657BC	0.0391ab	6.6cd
	Min	2,369.62	0.0018	0.37	0.18	0.0237	0.04
	Max	2,776.32	0.2947	0.64	1.00	0.0594	22.5
	SD	183.73	0.1204	0.09	0.37	0.0097	8.0

Note. Ma-Be: maize-bean; Ma-Be-Af: maize-bean-agroforestry; Ma-Be-Af-HVT: maize-bean-agroforestry-high value trees. Values in column followed by the same letter are not significantly (α = 0.05) different.

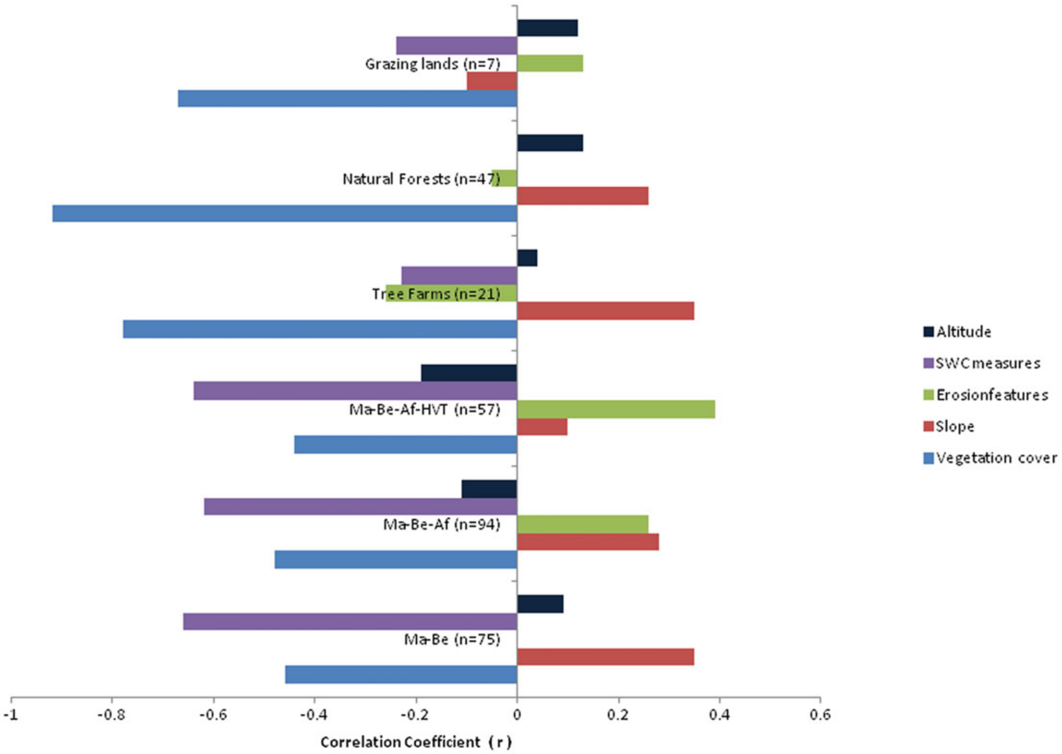

FIGURE 4 Correlation (*r*) between modelled annual soil loss and selected biophysical properties in major agro-ecosystems of Lushoto District, Tanzania [Colour figure can be viewed at wileyonlinelibrary.com]

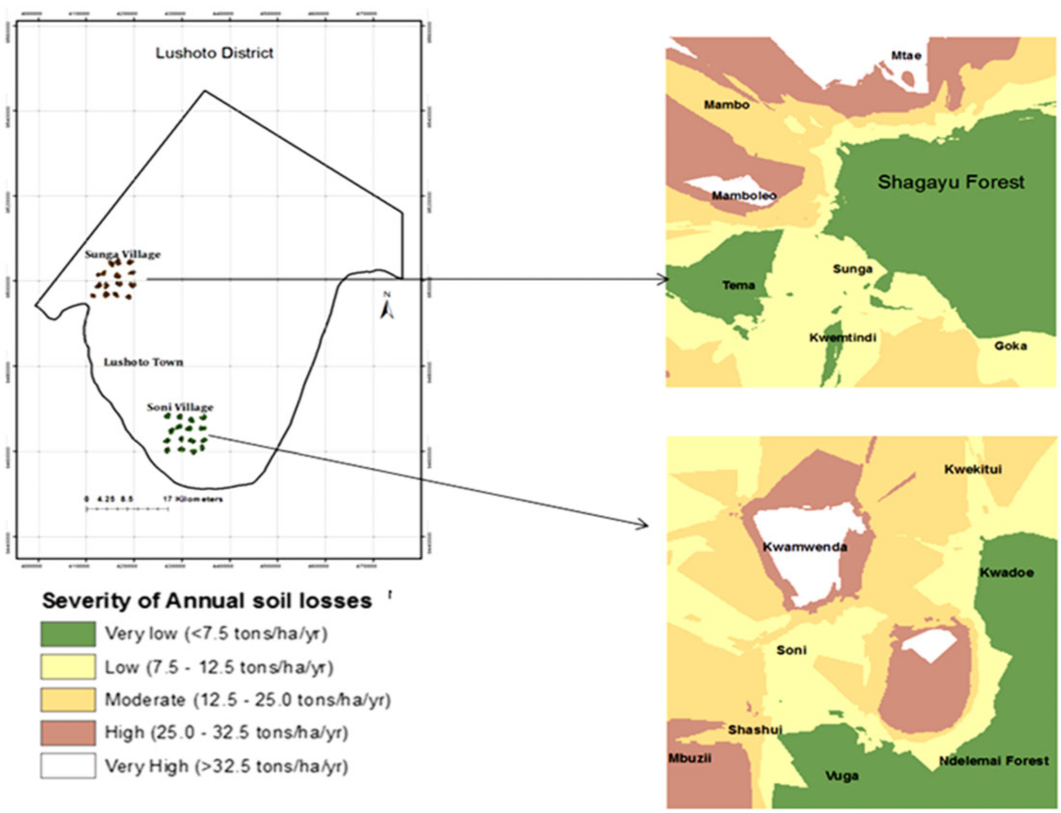

FIGURE 5 Severity of soil erosion in Soni and Sunga villages, Lushoto Tanzania [Colour figure can be viewed at wileyonlinelibrary.com]

combined proportion of natural forests observed in this study (15.95%, derived from Table 2) corresponds to the 14.7% forest cover remaining in the WUH in general, as reported by Newmark (1998). This similarity indicates the reliability of combining soil surveys, GIS, and modelling in mapping and delineating agro-ecosystems. Further, similarity of these proportions almost 20 years later indicates that by-laws for gazetting natural forests in Lushoto District are effective in stopping rampant deforestation.

Furthermore, our methodology of randomization of locations from which to collect data proved useful in this study. If these locations were sampled differently or with bias—as often happens in soil survey assessments—then potentially different proportions for each agro-ecosystem would have resulted. Differences observed in spatial extent, slopes, vegetation cover, and erosion features of agro-ecosystems (Tables 2 and 3) are related to varying geo-ecological conditions across the WUH (Ezaza, 1988) and the land use history in the area, whereby prime forests were allocated to farmers as agricultural land after independence of Tanganyika in the 1960s. Similarly, in New Hampshire (United States), historical land use disturbances also culminated with altitudinal arrangements of natural plant communities (Sperduto, Nichols, & Kimball, 2004). Variations (p < 0.05) observed in top/subsoil conditions in our study were expected because of movements of soil constituents across soil depths and nutrient recycling processes through defoliation and tillage practices (Winowiecki et al., 2016).

The predicted annual soil losses in the maize-bean agro-ecosystem (with an average of 28.3 t ha^{-1} yr^{-1}) are high to severe in most locations. These losses are higher than those reported from a nearby village of Kwalei (22.9 t ha^{-1} yr^{-1}) for maize fields (Tenge, Sterk, & Okoba, 2011), but similar (32.3 t ha^{-1} yr^{-1}) to erosion rates reported earlier in Lushoto for unconserved macro-contour lines for erosion control (Pfeiffer, 1990). These soil losses in maize-bean agro-ecosystem in our study were almost 18 times higher as natural erosion rates in nearby forests (Table 5). This implicates that tillage practices in the two villages enhance soil erosion, and that priority in soil conservation efforts should be directed to locations dominated with maize-bean agro-ecosystem (Figure 4). However, our soil loss estimates were lower than those reported from other agro-ecosystems in Uganda (Lufafa, Tenywa, Isabirye, Majaliwa, & Woomer, 2003), where an annual loss of 93 t ha^{-1} yr^{-1} in maize-legume agro-ecosystem was estimated. In Rwanda, soil losses around 41.5 t ha^{-1} yr^{-1} were recorded in maize-potatoes agro-ecosystems (Kagabo et al., 2013), and in Ethiopia, soil losses ranging from 0.4 to 88 t ha^{-1} were reported from maize-based smallholder agro-ecosystems (Tamene, Adimassu, Ellison, et al., 2017). Therefore, soil losses predicted in our study, though typical for East African Highlands, are generally lower than reported by other studies.

From our erosion modelling results, it can also be deduced that installation of 100 bench terraces or grass strips per ha would reduce soil losses by 7.8 t ha^{-1} yr^{-1}. Similarly, each 1% increase in surface slope would result into a 1.53% increase in soil loss, which is similar to what was reported from Ethiopia (Tamene, Adimassu, Aynekulu, & Yaekob, 2017). Our results also show that each 10% increase in vegetation cover eventually would reduce soil loss by 5.2 t ha^{-1} yr^{-1}. In Malawi, agro-forestry hedges on a 44% slope reduced soil losses by 97% within 6 years (Banda, Maghembe, Ngugi, & Chome, 1994) due to increased vegetative cover. From these results, it can be deduced that agro-ecosystems with conservation measures reduce soil losses by up to 37% and build organic carbon levels by at least 16% compared with unconserved agro-ecosystems (Tables 4 and 5). However, given that the model we used (USLE) is unable to predict concentrated linear soil erosion adequately (Renard et al., 1991), we recommend to continue evaluating soil losses in these agro-ecosystems, and monitor how erosion maps change in time and due to management considering. Furthermore, it would also be preferable to actually measure soil losses in some of these agro-ecosystems in order to establish any contradiction between modelled and observed soil losses, and to evaluate the accuracy of the models used.

5 | CONCLUSIONS

This study has demonstrated that when ground soil surveys are complimented with GIS and soil erosion modelling, it is possible to map spatial distribution, management practices, and soil erosion situations in smallholder agro-ecosystems. This is particularly interesting in circumstances in which such an assessment is impossible by means of satellite imagery and RS techniques alone. Soil erosion rates predicted by this approach are comparable and within range to those generated by alternative approaches in the region, which proves that the approach is useful. Furthermore, the results of the erosion modelling show that 40% of the areas under annual crops (maize and beans) have annual soil losses exceeding 25.5 t^{-1} ha^{-1}. These losses are nearly 18 times higher compared with those in nearby natural forests. Hence, agro-ecosystems with annual cropping practices are major contributors to soil erosion in the WUH and should receive priority in any soil conservation effort. Proper tillage and effective agronomic practices should therefore be promoted among smallholder farmers, alongside effective SWC measures.

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