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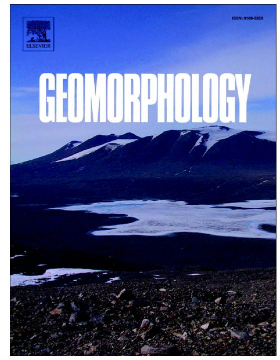

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Mapping soil erosion hotspots and assessing the potential impacts of land management practices in the highlands of Ethiopia

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ABSTRACT

An enormous effort is underway in Ethiopia to address soil erosion and restore overall land productivity. Modelling and participatory approaches can be used to delineate erosion hotspots, plan site- and context-specific interventions and assess their impacts. In this study, we employed a modelling interface developed based on the Revised Universal Soil Loss Equation adjusted by the sediment delivery ratio to map the spatial distribution of net soil loss and identify priority areas of intervention. Using the modelling interface, we also simulated the potential impacts of different soil and water conservation measures in reducing net soil loss. Model predictions showed that net soil loss in the study area ranges between 0.4 – 88 t ha^{-1} yr^{-1} with an average of 12 t ha^{-1} yr^{-1}. The dominant soil erosion hotspots were associated with steep slopes, gullies, communal grazing and cultivated areas. The average soil loss observed in this study is higher than the tolerable soil loss rate estimated for the highland of Ethiopia. The scenario analysis results showed that targeting hotspot areas where soil loss exceeds 10 t ha^{-1} yr^{-1}

could reduce net soil loss to the tolerable limit (less than 2 t ha^{-1} yr^{-1}). The spatial distribution of soil loss and the sediment yield reduction potential of different options provided essential information to guide prioritization and targeting. In addition, the results can help promoting awareness within the local community of the severity of the soil erosion problem and the potential of management interventions. Future work should include cost-benefit and tradeoff analyses of the various management options for achieving a given level of erosion reduction.

Key words: Land degradation; sediment delivery ratio, graphical user interface; soil erosion management scenario, Basona district

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1. Introduction

The highlands of Ethiopia are subject to a high level of land degradation (Bojo and Cassels, 1995; Jolejole-Foreman et al., 2012) through deforestation, soil erosion, and nutrient mining (Sonneveld and Keyzer, 2003). Soil erosion rates in the cultivated highlands of Ethiopia can reach over 130 t ha^{-1} yr^{-1} (Berry, 2009). The direct cost of soil loss and its essential nutrients due to unsustainable land management reaches $106 million annually (Bojo and Cassels, 1995; Berry, 2009). Due to excessive soil loss rates in the uplands, a majority of water harvesting schemes have struggled to meet their intended potential in providing irrigation water to supplement the highly unreliable rainfall (Tamene and Vlek, 2007). Soil erosion can also cause sedimentation of hydropower dams and result in significant economic losses due to frequent power cuts. High erosion and sedimentation can also undermine the overall provision of ecosystem functions and services of landscapes (Fu et al., 2011; Mekuria et al., 2011; Yimer et al., 2015).

To minimize the impacts of erosion, there is a need to prevent further degradation through natural recovery and restoration strategies such as exclosures, reforestation, as well as other soil and water conservation (SWC) interventions (Mekuria et al., 2011; Schwilch et al., 2013; Mekonnen et al., 2015). Since erosion shows spatial variability across the landscape, it is not economically feasible or technically possible to conserve all areas experiencing soil loss. As a result, it is necessary to identify critical areas of erosion that require priority management intervention. Spatially distributed erosion models can be used to identify erosion hotspots and simulate the potential impacts of different land management options (Arnold et al., 1998; Mitasova et al., 2001; Tamene et al., 2014).

A wide range of models are available to estimate soil erosion at different scales (e.g., the Universal Soil Loss Equation (Wischmeier and Smith, 1978); Water Erosion Prediction Project (Nearing et al., 1989); Revised Universal Soil Loss Equation (Renard et al., 1997); the European Soil Erosion Model

(Morgan et al., 1998); and the Soil and Water Assessment Tool (Arnold et al., 1998; Neitsch et al., 2005) to name a few). However, the application of some of the models, especially in developing regions, renders some challenges such as data availability at the desired scale, resolution and accuracy. Despite the fact that some of the erosion models have relatively modest data requirements, they are not flexible enough to be used by local institutions and stakeholders (Tamene et al., 2014). To guide planning and decision-making with regards to identifying critical areas of intervention and target appropriate management interventions, it will be useful to develop a modeling framework that can be implemented at the local level (Vervoort et al., 2010; McIntosh et al., 2011). The availability of such tools is essential in order to better equip local stakeholders to plan land management interventions and estimate their potential impacts (Tamene et al., 2014). Enabling stakeholders to compare the results of different options and choose the ones they think are acceptable for their conditions is also essential to facilitate technology adoption and out-scaling (Frost et al., 2006).

In this study, we used a modelling framework developed to assess the severity of soil loss and evaluate the impacts of different SWC practices in reducing sediment yield in an example watershed in the highlands of Ethiopia. Specifically, the study aims to: (a) identify major areas of concern (hotspots) for management interventions; (b) identify site-specific management practices to target the priority areas in a participatory manner; (c) simulate and analyze the role of different land management options in reducing soil loss; and (d) evaluate the usefulness of the modelling results to guide planning and decision-making in the context of current land management initiatives in Ethiopia.

2. Study area and site characteristics

The study was conducted in the Laygeda watershed of the Basona district in the highlands of Ethiopia. The watershed is located in the Gudo Beret and Adisge Kebeles (a kebele is the smallest official administrative unit in Ethiopia), about 160 km northeast of Addis Ababa (Fig. 1). The area of the watershed is about 15,250 ha and features an elevation range of 2500 – 4000 m above sea level. Temperatures range from as low as -2°C during autumn/winter to over 25°C during spring/summer seasons. The area receives about 1100 mm per annum rainfall, mainly during two seasons: up to 1000 mm falls during the summer season (July-September) and up to 200 mm falls during the short rain period (March-May).

The dominant farming system of the study area is mixed crop-livestock production. Cultivated areas and woodlots (dominated by eucalyptus) are the major land use/cover types. During the long rains, wheat, barley, and faba bean are the dominant crops. Barley also grows during the short rains mostly in the uplands of the watershed. While Eucalyptus (*Eucalyptus globulus*) is the major tree species in the area, Juniper (*Juniperus procera*) in the highlands and guassa grass (*Festuca spp.*) in the more extreme upland areas are also common tree and grass species.

[Figure 1 around here please]

3. Methodology

3.1. Model selection and parameterization

In this study, we used the Revised Universal Soil Loss Equation (RUSLE) model (Renard et al., 1997) to estimate soil loss and map its spatial variability. This model has been selected because it requires minimal data and has a low computational cost, rendering it more suitable for data scarce regions (Garg and Jothiprakash, 2012; Chowdary et al., 2013). In addition, some of the key model parameters have been calibrated for Ethiopian conditions (e.g., Hurni, 1985). The RUSLE is defined as (Renard et al., 1997):

$$RUSLE\ (t\ ha^{-1} year^{-1}) = RxKxLSxCxP \qquad \text{(eq.1)}$$

where, R = rainfall erosivity (MJ mm ha^{-1} h^{-1} yr^{-1}); K = soil erodibility (t ha h ha^{-1} MJ^{-1} mm^{-1}); LS = slope length-steepness (-); C = land use/cover (-); and P = conservation/management (-) factors.

The parameters of the RUSLE model and the methods of their extraction are discussed below.

3.1.1. The slope-length (LS) factor

The slope steepness and slope length factors often referred to as ‘‘LS’’ factor represent the role of terrain steepness and its configuration on soil movement across landscapes, fields or plots. Some studies have shown that *upslope contributing area* (that is, the area upslope of a given pixel which contributes runoff to that pixel) better represents the impact of topography on soil erosion than length of slope when applied at a landscape scale (Moore and Burch, 1986; Desmet and Govers, 1996a, b). The upslope contributing area is also essential to consider the impact of flow convergence on soil erosion and better reflects the impact of concentrated flow on increased erosion especially in hilly areas (Mitasova et al., 1996). The LS-factor at the landscape scale can be estimated based on the unit contributing area and slope steepness (Moore and Burch, 1986; Moore et al., 1991):

$$LS = (m+1)\left[\frac{A_s}{22.13}\right]^m \left[\frac{\sin\beta}{0.0896}\right]^n \qquad \text{(eq.2)}$$

where, m (0.4 – 0.56) and n (1.2 – 1.3) are slope length and angle coefficients; A_s is the specific upslope contributing area per unit length of contour; and β is the local slope gradient (degrees).

The unit contributing area (A_s) can be calculated by multiplying a flow accumulation grid with the cell size (Moore and Burch, 1986; Mitasova et al., 1996):

$$A_s = \frac{1}{b_i}\sum_{i}^{N} a_i \mu_i \qquad \text{(eq.3)}$$

where, a_i is the area of the i^{th} grid cell; b is the contour width of the i^{th} cell (approximated by pixel resolution); μ_i is the weight depending upon the runoff generating mechanism and infiltration rates; N is the number of grid cells draining into grid cell i. In this study we used μ=1 assuming that rainfall excess runoff is generated uniformly over the landscape (Moore et al., 1988; Mitasova et al., 1996; Fernandez et al., 2003).

To derive the LS-factor, the recently released 30 m resolution Shuttle Radar Topography Mission (SRTM) digital elevation model (DEM) was used. The DEM was first pre-processed to fill spurious depressions in order to route runoff and associated sediment to the catchment outlet.

3.1.2. Rainfall erosivity (R) factor

The erosivity of rainfall events (*R*-factor) is defined as the product of kinetic energy and the maximum 30-min intensity (Wischmeier and Smith, 1978; Renard et al., 1997). In areas where rainfall intensity data is not commonly available, attempts have been made to establish a relationship between mean annual or monthly rainfall and rainfall intensity to estimate the *R*-factor (Renard and Freimund, 1994; Le Roux et al., 2008; Xin et al., 2010). For the Ethiopian condition, we used an empirical equation established by Hurni (1985) to derive *R*-factor from mean annual rainfall data:

$$R = 0.36 * MAP + 47.6 \qquad \text{(eq. 4)}$$

where, R is the rainfall erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}) and *MAP* is mean annual precipitation (mm).

We applied Eq. (4) to derive the *R*-factor from a rainfall surface map generated using rainfall data of over 200 meteorological stations within Ethiopia in combination with the 30 m DEM following the approaches suggested by (Huang and Tiesong, 2009). Despite the fact that our study area is relatively small, we preferred to use a 'spatially distributed' rainfall surface map in order to pick up on potential variations across the 1500 m elevation gradient of the watershed.

3.1.3. Soil erodibility (K) factor

While the ability of rainfall energy to 'dislodge' soil particles is key to soil erosion, the propensity of soil materials to be dislodged and transported with a given energy level is also significant. The *K*-factor expresses the property of soil particles that govern its resistance to detachment and transport by water

(Renard et al., 1997). The common soil properties that affect soil particles' ability to detach and erode include texture, organic matter, structure and permeability (Wischmeier et al., 1971; Renard et al., 1997). In this study, we used the set of equations developed by Auerswald et al. (2014, 2016) to derive *K*-factor values from basic soil property data. A digital soil map generated for the area following the approach developed by the International Soil Reference and Information Center (ISRIC) (Hengl et al., 2015) was used to extract the necessary parameters and derive the *K*-factor.

The approach developed by Auerswald et al. (2014, 2016) involves four steps. Despite the estimation that stone mulches can reduce erodibility by up to 10% (Wischmeier et al., 1971), we did not include this adjustment as provided by Auerswald et al. (2014, 2016) due to a lack of data for the percent stone cover on the soil surface. For this reason, Eq. (5) was used in our study:

$$K1 = 2.77 * 10^{-5} * \left(fSi + vfSa * (100 - fCl)\right)^{1.14} \qquad for\ fSi + vfSa < 70\%$$

$$K1 = 1.75 * 10^{-5} * \left(fSi + vfSa * (100 - fCl)\right)^{1.14} * 0.0024 * fSi + vfSa + 0.16 \qquad for\ fSi + vfSa > 70\%$$

$$K2 = \frac{12 - fOM}{10} \qquad for\ fOM < 4\%$$

$$K2 = 0.8 \qquad for\ fOM > 4\%$$

$$K3 = K1 * K2 + 0.043 * (A - 2) + 0.033 * (P - 3) \qquad for\ K1 * K2 > 0.2$$

$$K3 = 0.091 - 0.34 * K1 * K2 + 1.79 * (K1 - K2)^2 + 0.24 * K1 * K2 * A + 0.033 * (P - 3) \quad for\ K1 * K2 < 0.2$$

(eq. 5)

where, fSi is the percentage of silt, fSi + vfSa is the combined percentage of silt and very fine sand, fCl is the percentage of clay, fOM is the percentage of soil organic matter, P is the permeability index, and A is the structure code.

Since data for "very fine sand" was not explicitly available, it was estimated as 20% of the total sand fraction (Panagos et al., 2014). Permeability index corresponds to the FAO drainage classes whereby seven levels of drainage are distinguished ranging from "excessively drained" to "very poorly drained" (FAO, 1990). Structure code was determined from packing density (PD) which was estimated based on King et al. (1995):

$$PD = \rho b + fCl \qquad (eq. 6)$$

where, ρb is bulk density in g cm^3 and fCl is the percentage of clay in the top 30 cm. Structure classes were then assigned based on the pedotransfer rule shown in Table 1 **(**King et al., 1995).

[Table 1 around here please]

3.1.4. Surface cover (C) factor

Surface cover is one of the key components playing a significant role in determining the rate of soil loss (Cerdà, 1997). In general, areas covered with vegetation have a high surface roughness, which can increase infiltration and reduce runoff, while cultivated/bare soil areas have a low surface roughness resulting in high runoff (Cerdà, 1997; Bull et al., 2003). To account for such differences on soil loss, the *C*-factor was incorporated in soil erosion models (Wischmeier and Smith, 1978). In this study, *C*-factor values derived from land use/cover (LUC) maps were used. The LUC maps were derived from high resolution (0.5 m) satellite images acquired from the Pleiades system (http://www.intelligence-airbusds.com/pleiades/). The C-factor values were derived from the LUC map using class values calibrated by Hurni (1986) for Ethiopian conditions (Table 2).

3.1.5. Support practice (P) factor

The severity of erosion in an area is dictated by the degree of conservation practices in place or the magnitude of human influence exerted on it. The *P*-factor is the ratio of the soil loss expected for a certain soil conservation practice to that which occurs under up- and down-slope plowing (Wischmeier and Smith, 1978). The *P*-factor values are generally derived from spatially distributed land conservation or management data. Recent soil and water conservation campaigns in Ethiopia have targeted different watersheds, resulting in marked differences across these landscapes. In this study, we conducted participatory transect walks to evaluate the extent and condition of conservation practices. Considering the shape of the watershed, the study site was 'partitioned' into three transects which were walked over the course of four days. Three of the co-authors accompanied by eight farmers (selected based on experience and knowledge of the area) participated during the transect walks. The resulting observations were used to inform and derive *P*-factor values for corresponding land use/cover types based on Hurni (1985) (Table 2).

[Table 2 around here please]

3.1.6. Estimating sediment delivery ratio to adjust gross soil loss

The RUSLE is designed to predict total annual soil loss at various scales (pixel, field, or watershed), but does not consider the potential for suspended materials to be deposited at some point downslope from their origin. In fact, not all of the soil eroded from the upper parts of a watershed will be delivered to an outlet (Walling, 1983). Depending on surface cover and terrain characteristics, much of the material can be re-deposited at locations where the momentum of runoff water is insufficient to further carry the eroded material downslope or downstream inside a channel. Generally, the potential of intermediate deposition increases as the watershed area increases because there are more opportunities for eroded sediment to settle (Boyce, 1975). To account for deposition, we used the sediment delivery ratio (SDR), the fraction of eroded soil from a defined area that successfully reaches a water channel (Walling, 1983; Stefano et al., 2005). According to Stefano et al. (2005), SDR can be calculated as:

$$SDR_i = \exp(-b * \frac{L_i}{R_i S_i^{1/2}}) \qquad \text{(eq.7)}$$

Where, β is a routing coefficient; L_i is the length of segment *i* in the flow path and is equal to the length of the side or diagonal of a cell depending on the flow direction in the cell; R_i is a coefficient based on surface roughness characteristics; and S_i is the slope gradient (m m^{-1}).

The β coefficient represents a 'watershed specific' parameter to characterize the effects due to roughness and runoff along the hydrologic path and primarily depends on watershed morphological parameters (Ferro and Porto, 2000). Some studies assumed β values of unity when estimating SDR using Eq. (7) (Jain and Kothyari, 2000; Bhattarai and Dutta, 2007; Fu et al., 2006; Mutua et al., 2006). Jain and Kothyari (2000) showed that the computed values of sediment yield were not more than 10% in any of the storm events for a large range of β values. Ferro et al. (2003) calibrated β coefficients for six catchments corresponding to different expressions of the topographic factor and identified ranges that are very close to β values calculated by a sediment balance equation. In this study, we used a β coefficient of 0.0051 estimated by Ferro et al. (2003) corresponding to the topographic factor expression of Moore and Burch (1986) which is used in our LS-factor estimation.

The surface roughness coefficient (R_i) used in the calculation of SDR per pixel is estimated from land use/cover types, as suggested by Maidment et al. (1996), Jain and Kothyari (2000), Fernandez et al. (2003), Stefano et al. (2005), and Mutua et al. (2006). Table 2 shows the estimated values for the velocity coefficient adopted in this study.

3.2. Identification of the major areas of concern and drivers of soil erosion

About 20 farmers who exhibited a broader knowledge of the watershed were involved to identify the major sites and causes of soil erosion in the study watershed. During the discussion, communities identified gullies to be the main drivers of soil loss. In order to target these areas during our scenario analysis, it was necessary to demarcate gullies on our map. Accordingly, the potential location of gullies was delineated (Fig. 2) based on Thorne et al. (1986) and Moore et al. (1988):

$$A_s tan\beta > 18 \quad \text{(eq.8)}$$

$$ln\left(\frac{A_s}{tan\beta}\right) > 6.8 \quad \text{(eq.9)}$$

where, A_s = unit contributing area ($m^2\ m^{-1}$); $tan\ \beta$ = local slope (tan). Gullies can be located when the above two conditions are met.

In addition to gullies, farmers identified terrain slope as an important driver of soil erosion. It was thus necessary to define a slope threshold beyond which soil erosion is likely to be serious and thus where conservation practices should be in place. In this study, we defined 25% slope to be the maximum slope threshold beyond which SWC measures must be implemented (Desta at al., 2005a). Upon seeing these high slope areas on maps and on the ground, we believe that those locations indeed are sensitive to erosion and should be targeted for conservation.

Discussion with communities and field visit indicated that degraded communal and cultivated areas which are generally characterized by poor surface cover are also sensitive to and/or are experiencing high soil loss. In order to accommodate these areas in our scenario analysis, we used the concept of the ‘tolerable’ soil loss rate (amount of soil loss that is within an acceptable range considering severity of erosion, soil thickness and soil formation rate) to define the hotspot areas of soil erosion. Considering the range of 2-18 t $ha^{-1}\ yr^{-1}$ tolerable soil loss rate suggested for Ethiopia (Hurni, 1986), we decided to use the a threshold figure of 10 t $ha^{-1}\ yr^{-1}$ to be areas designated as hotspots and thus requiring priority management interventions.

[Figure 2 around here please]

3.3. *Modeling sediment yield and simulating the impacts of management options*

To facilitate soil erosion estimation in data scarce regions and ease of use by local stakeholders, it was essential to develop a modelling framework that is flexible for data input, adjustment of erosion factors or coefficients, and for the simulation of the potential impacts of different interventions in reducing soil loss. In this study, we used a model with a graphical interface developed using the RUSLE adjusted for the SDR to estimate net soil loss and assess the impacts of different SWC practices (Tamene et al., 2014).

For the scenario analysis, it was essential to identify the soil erosion factors that can be influenced through management practices. Among the five factors used in RUSLE, two factors can be directly modified through human interventions to reduce soil loss; for example, surface cover (C) can be altered through re-vegetation of an area, and management (P) through use of an alternative cropping system. The influence of slope length (LS) on runoff and soil loss can also be affected through the introduction of terracing. Soil erodibility can be indirectly altered in the long-term through management regimes or land uses that promote an accumulation of organic matter. Although it will not be possible to directly influence rainfall through SWC options at local level, it could be possible to adjust the rainfall erosivity component through changing rainfall amounts. Considering these, three of the major RUSLE soil erosion components (*C*-, *P*-, and *R*- factors) were the basis of the scenario analysis in this study. From experience and based on discussion with farmers, the most feasible interventions for tackling soil loss include improving surface cover (planting trees, exclosures, etc.) and constructing terraces and buffering gullies (e.g., Cerdà, 1997, Cerdà et al., 2009; Tesfahunegn et al., 2012; Adimassu et al., 2017).

When running simulation, users (practitioners, planners, extension workers, researchers, and students) can first decide *where* to intervene (steep slopes, gullies, hotspots, or a combination of those), and then assign relevant interventions (*C*- and/or *P*-factors) corresponding to the specific issues (drivers) of relevance to the study area. In the case of the *R*-factor, the average rainfall can be updated to consider the possible impact of a change in mean annual rainfall. When 'rainfall amount' is changed in the model, the *R*-factor will update itself in relation to the empirical relationship defined in Eq. (4).

Once model variables are imported and coefficients are adjusted, the current (status quo) soil erosion rate can be estimated considering 'business as usual' practices. The next step is then to decide where interventions should be targeted (steep slopes, gullies or erosion hotspots). If the option is to target slopes, the modelling framework allows users to choose a threshold slope beyond which interventions should be applied. Because the definition of steep slopes in the context of soil loss could differ from place to place, a range of possibilities (from 5-50%) is allowed in the modelling framework. In the default example, areas with slopes greater than 25% are suggested to be enclosed and assigned a *C*-factor value of 0.01. The value of 0.01 is assigned because it is unlikely that exclosures will turn into dense forest in the short term (Tamene et al., 2014). In addition, a simulation can be run for the terracing of steep slope areas (*P*-factor = 0.6). The tool also provides an option to identify gullies (using Eqs. 8 and 9) and assign management options. In this study, the gullies and a buffer zone 5-20 m from the gully center are suggested to be protected through terracing and planting of deep-rooting grasses (Verstraeten and Poesen, 2002; Borin et al., 2005). Although values of 0.6 for the *P*-factor and 0.01 for the *C*-factor are used as post-intervention defaults in the tool, the model also offers the possibility to choose from a range of *P*- and *C*-factor values when more information is known concerning local conditions and the effectiveness of the proposed management efforts. A fourth option targets erosion hotspots exceeding 10 t ha^{-1} yr^{-1} soil loss; the tolerable amount defined for this study considering the range of valued provided in Hurni (1986). Although a threshold of 10 t ha^{-1} yr^{-1} is used to identify erosion hotspots in the model, users have the option to target areas based on their local conditions. For instance, a smaller tolerable soil loss rate might be assumed for a site similar to the one in our study but having shallower soils. The framework also enables experimenting with different levels of soil loss if the tolerable amount is not known. For the hotspots, a broad range of *C*- and *P*-factor values can be tested, although values of 0.01 for *C* and 0.6 for *P* are suggested in the model. Users can adjust the *C*- and/or *P*-factor values for their local context based on their experiences and the performances of certain SWC practices measures in the past. Finally, the model can be used to simulate the impacts of integrated SWC interventions targeting steep slopes, gullies and soil erosion hotspots.

3.4. *Evaluation of model results*

Model evaluation is essential to gain an overall understanding of its performance and gauge its applicability. Different approaches can be used to assess the accuracy of the soil loss prediction (Sonneveld et al., 2011). In this study, we compared our soil loss estimation with others conducted in the region. However, information related to the rate of soil loss alone will not offer much to guide planning and targeting. It is thus important to assess how well the spatial distribution of soil loss is predicted as this helps prioritizing intervention areas. Due to a lack of spatially distributed information to validate model performance, we employed visual assessment using experts' knowledge of the area and focused field visits to qualitatively evaluate the relative erosion risk of different landscapes.

4. Results and Discussion

4.1. Soil erosion rate and its spatial distribution

The gross soil loss of the study site estimated using the RUSLE model ranged from 1 to 862 t ha^{-1} yr^{-1} with an average of 48 t ha^{-1} yr^{-1}. The majority of the areas experiencing high soil loss are associated with steep cliffs, gully sides and degraded quarry areas. The average SDR estimated using Eq. (7) was about 26%. When we observe the spatial distribution, we see that areas close to gullies or streams exhibited SDR values close to 100%, which shows that most of the sediment in these cells will be delivered to their respective outlets. However, the majority of the study area showed low to moderate SDR values due to mainly flat terrain and dense surface cover in some parts (Fig. 3). The average SDR of 26% in this study is higher than the average SDR of about 6% observed by Muleta et al. (2006) for a 4755 ha watershed. The SDR observed in this study can be labeled as moderate considering the relatively expansive network of collapsing gullies that serve both as sediment sources and agents of transport.

The net soil loss (after adjustment for the SDR) based on model prediction in the study area ranges from 0.4 to 88 t ha^{-1} yr^{-1}. When areas of very high soil loss, which cover less than 0.5% of the watershed area are excluded, maximum soil loss declines to about 66 t ha^{-1} yr^{-1}. The average annual net soil loss estimated for the study watershed was 12.3 t ha^{-1} yr^{-1}.

(Fig. 3 around here please)

About 11,000 ha (70%) of the study area experienced an average soil loss rate of less than 1 t ha^{-1} yr^{-1} while about 1,444 ha (10%) of the study area experienced a soil loss rate of more than 15 t ha^{-1} yr^{-1}. Areas with slopes greater than 35% are characterized by an average soil loss rates of about 22 t ha^{-1} yr^{-1}. This is more pronounced on areas where surface cover is sparse. On relatively flat areas of less than 8% slope, the average soil loss rate was generally less than 3 t ha^{-1} yr^{-1}. This is mainly because the energy of water flow dissipates and deposition dominates in flat areas. However, lowland areas are still subject to significant soil losses due to gullies, which generally tend to form at lower slopes where concentrated flows occur (Poesen et al., 2003). In addition, flat areas are dominantly used for cultivation practices which can render the land more vulnerable to soil erosion. In terms of land use, the guassa (*Festuca spp.*) grassland, which covers about 350 ha and is generally found on the extreme upper parts of the watershed, experienced the lowest soil loss (less than 1 t ha^{-1} yr^{-1}). The area is also generally characterized by low slope. Cultivated areas exhibited an average net soil loss rate of around 23 t ha^{-1} yr^{-1}. This rate is significantly higher than the maximum tolerable limit though lower than some estimates of soil erosion for cultivated areas in Ethiopia.

Generally, the observed soil loss in the majority of the study area is not as severe as those of related studies due to a combination of different factors. The upper part of the watershed is covered with dense grass which significantly reduces soil loss. In addition, parts of the relatively steep areas are covered with dense eucalyptus woodlots that can reduce the energy of water flow and its erosive power (e.g., Cerda, 1997, Cerda et al., 2009). In a majority of the watershed, various 'traditional' and modern conservation structures are in place. The predominant interventions include terraces and trenches in some cases supplemented with biological SWC measures such as tree lucerne (*Chamaecytisus palmensis*). This could be one of the reasons why soil loss from cultivated areas in our study areas was lower than that estimated by Hurni (1993), which was about 42 t ha^{-1} yr^{-1}. In the Tigray region of the Ethiopian highlands, Desta et al. (2005b) estimated a soil loss rate of 20 t ha^{-1} yr^{-1} from cultivated fields with stone bunds, which is closer to the estimate in our study. In addition to stone bunds, the traditional field boundaries (between parcels of different owners) can dissipate the energy of water flow, thereby reducing soil detachment and movement (Nyssen et al., 2009).

Despite the fact that the range and average soil losses help to give an indication of the severity of the problem, they are not very useful for targeted planning and decision-making (Mitas and Mitasova, 1998). Further, estimated values will have wide uncertainty due to the various datasets used in estimating soil erosion. It is thus beneficial to focus on the overall soil erosion risk and the spatial patterns to identify areas that need priority attention. Fig. 4 shows the spatial variations in soil erosion, which can guide where interventions should be prioritized. Using the map it is possible to query for different levels of soil loss and plan to target those areas with suitable conservation measures. Generally, areas with dense surface cover experienced low soil loss while steep slopes and sparsely covered areas are characterized by high soil loss.

[Figure 4 around here please]

4.2. Comparison of model results with other studies in the country

The average soil loss rate observed in this study was compared with other 'model-based' estimates made in the country. Nyssen (1997) applied the Universal Soil Loss Equation (USLE) for a study area in Doga Temben (Tigray region of Ethiopia) and reported a soil loss rate of about 11 t ha^{-1} yr^{-1}. Senti et al.

(2014) applied the Modified Universal Soil Loss Equation (MUSLE) in eastern Ethiopia and observed a soil loss rate of about 24 t ha^{-1} yr^{-1}. Another study using the RUSLE in the Guang watershed of the Blue Nile Basin of Ethiopia reported a soil loss rate of about 25 t ha^{-1} yr^{-1} (Ayalew and Selassie, 2015). These studies which employed USLE/MUSLE/RUSLE considered the gross soil loss without adjusting for the sediment deliver ratio – a possible reason for the relatively higher soil losses estimated in some cases. In addition, the study sites covered small areas (up to 5000 ha) compared to our watershed (over 15,000 ha). This difference would likely cause a variation in net soil loss since larger areas have more intermittent deposition, leading to a lower sediment yield (e.g., Walling, 1983). A study by Setegn et al. (2010) applied the Soil and Water Assessment Tool (SWAT) at the gauged Anjeni watershed with an area of 113 ha and estimated an average annual sediment yield of 24 t ha^{-1} yr^{-1}. A long-term average soil loss estimate using SWAT for a watershed of about 106,000 ha showed a soil loss rate of about 4.3 t ha^{-1} yr^{-1} (Tibebe and Bewket, 2011). A study by Megersa (2014) using the Agricultural Non-Point Source (AGNPS) pollution model reported an average soil loss rate of 17 t ha^{-1} yr^{-1} for a study area in eastern Harerghe. A recent study by Hurni et al. (2015) based on the Unit Stream Power-based Erosion-Deposition (USPED) model also estimated the national average soil loss rate at about 18 t ha^{-1} yr^{-1}. For a smaller sub-catchment within our watershed (2,500 ha), Ellison (2016) estimated an average soil loss of about 20 t ha^{-1} yr^{-1} using the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) model. Gauging stations installed to monitor runoff and sediment yield within two sub-catchments of the study watershed showed an average suspended sediment yield of about 5 t ha^{-1} yr^{-1}.

The above results generally show that the SDR-adjusted RUSLE model result in this study is in agreement with other studies in the country. However, strict comparisons are rarely possible because of variations in the methods employed and their scale of application. It is thus essential to be cautious when validating model results, especially considering quantitative rates.

Field observation and discussion with communities generally showed that the spatial pattern of soil loss observed in this study (Fig. 4) confirm the conditions on the ground.

4.3. Scenario analysis of land management options

Fig. 5 shows the model graphical interface designed to facilitate soil erosion prediction and simulate the impacts of site-specific SWC interventions. The interface has various features to import (Feature 1), display (Feature 2) and inspect (Feature 3) the various datasets. It also enables the user to choose from a range of erosion factors and adjust coefficients (Feature 4) considering local conditions. In the *LS*-factor calculation (Eq. 2), values of m=1.0–1.6 and n=1.0–1.3 are used to calibrate the effects of slope length and steepness; values of m=1.6 and n=1.3 for rill erosion and m=1.0 and n=1.0 for sheet erosion are recommended (Warren et al., 2005; Liu et al., 2007; Rodriguez and Suarez, 2012). The m and n exponents can be calibrated if data are available for a specific prevailing type of flow or soil conditions (Mitasova et al., 1999, 2001; Leh et al., 2011). With flexibility in the interface, it is possible to experiment with different coefficients and observe the magnitude of change in the *LS*-factor and the associated effects in soil loss. The interface also offers ability to predict the potential locations of gullies, which can be used to simulate impacts of management options. This is useful because gullies are major causes and sources of erosion (Steegen et al., 2000; Poesen at al., 2003; Tamene and Vlek, 2007). After selecting the areas to be targeted, users can choose the respective management options to target each of the identified 'priority areas' (Features 5 and 6). Results of each scenario can be viewed in the form of descriptive statistics (minimum, maximum, average, and standard deviation) (Feature 8), map (Feature 2), as well as plot (Features 7 & 9). The framework also allows the results of the different scenarios to be exported (Feature 10).

[Figure 5 around here please]

Based on the modelling framework described above, an ex-ante analysis was conducted to assess the impacts of conservation practices in reducing soil erosion. The results in Fig. 6 show that soil erosion can be reduced from about 12 t ha^{-1} yr^{-1} to an average of about 7 t ha^{-1} yr^{-1} when conservation measures target only those areas with slopes >25%. Based on this, net soil loss would be reduced by about 38% by implementing conservation measures targeting steep slope areas. This supports the results of a study by Schmidt et al. (2014) in northwestern Ethiopia which showed that terraces on mid- and high-sloped areas can significantly reduce runoff and sediment yield. However, the amount of soil loss reduction due to conservation of relatively steep areas is not very high in relation to the total area that would need to be targeted. This is because areas with slopes steeper than 25% are not necessarily the major sediment sources in the study area. Although some of the steep cliffs exhibit extreme soil losses, they cover a very small area. In addition, it is important to note that areas with steep slopes are generally given priority when conservation efforts are planned and some of those areas have already been treated with terraces. Field visits and discussions with communities also revealed that some of the steep slopes have better vegetation cover (dense bushes, shrubs) because they are less accessible to humans and livestock.

When management measures targeting the gullies are introduced (with check-dams and biological measures), the net soil loss drops slightly, reaching about 10 t ha^{-1} yr^{-1} (~16% reduction). Targeting gully channels alone does not reduce soil loss significantly because it is not the gully bed that accelerates soil loss, but rather the gully banks supply a majority of soil for erosion. In line with this, net soil loss reduction improves (to about a 26% reduction) when gullies and their 20-m buffer are conserved (using check-dams and grass strips). By integrating management of steep slopes and gullies, net soil loss can be reduced by about 50% (Fig. 6).

[Figure 6 around here please]

Another option simulated is conservation of areas experiencing soil loss of more than 10 t ha^{-1} yr^{-1}. In this case, the overall soil erosion reduction is very significant (reaching an average net loss of 2 t ha^{-1} yr^{-1}) with a reduction of 83%. When conservation measures are simulated to target steep slopes, gully areas, and hotspots losing more than 10 t ha^{-1} yr^{-1}, the sediment yield drops to about 1.5 t ha^{-1} yr^{-1} (~88% decrease). A study in northern Ethiopia estimated a sediment yield reduction of about 78% due to integrated soil and water conservation measures (Nyssen et al., 2009). A review of the impacts of SWC measures on runoff, soil loss and crop yield by Adimassu et al. (2017) showed a soil loss reduction of about 88% due to stone bunds. An ongoing study by Yaekob et al. (in preparation) based on sediment yield measurements at a gauging station in a sub-catchment within the study area also points to an 80% sediment yield reduction for sub-catchments treated with terraces, trenches and biological measures. These observations show that site-specific integrated landscape management targeting sensitive areas of soil erosion and prescribing suitable management options can reduce soil erosion to below the tolerable limit.

Considering the fact that precision SWC is about applying conservation practices at the right place and at the right time, efforts should focus on targeting areas that produce the optimum benefit (Ellison, 2016). Based on Fig. 7, we can see that the spatial distribution of areas requiring interventions vary considering the basis of our prioritization (driver of erosion). By integrating such a map with land use/cover types, terrain slope and other biophysical and socio-economic conditions we can determine both the types of areas to be affected as well as the resources needed. It is important to note that although the terrain of the study area is rather elongated, its sediment delivery efficiency is relatively high because the various sub-catchments are well-connected with small streams and a network of gullies. It is thus necessary to develop methods to map and predict ephemeral and major gullies so that management plans can target those areas which act as major sediment sources and facilitate sediment delivery (Poesen et al., 1999; Nachtergaele et al., 2002; Koco, 2011).

[Figure 7 around here please]

The different simulations were run and results were discussed at a meeting where local administrators and lead farmers were present. The exercise stimulated debate by farmers about the performance of the various SWC interventions. The modelling tool generally increased awareness of the need for collective action among farmers and confirmed local perceptions as to the level of erosion that threatens their landscape. Based on the simulation results, farmers showed considerable concern about the areas where cultivation and grazing would be 'affected', especially when exclosures were suggested to protect the land from direct human and livestock interference. For instance, to achieve a soil loss reduction of about 50%, there is a need to take more than 3000 ha of land out of cultivation and prohibit direct interference by humans and livestock. This may not be a plausible alternative for the farmers whose average current land holding is less than 1 ha. The exclosure of some grazing areas (which, in total, account for 15% of the watershed) could also have an impact on livestock, so there is a need to develop alternative feed sources before implementing these interventions. With the introduction of physical measures such as stone bunds on cropland, some land area is inevitably taken out of cultivation, thereby impacting crop yield as demonstrated in different studies (e.g., Kato et al., 2011; Kassie et al., 2009; Adimassu et al., 2014). It is thus essential to conduct detailed cost-benefit and tradeoff analyses to determine which options and locations would give the greatest return on investment.

5. Conclusion

Soil erosion is a major problem affecting livelihoods and food security in Ethiopia. Understanding its magnitude, drivers and the critical areas affected can help us design suitable management interventions. The use of modelling tools that require optimum data and can easily be applied by local stakeholders can be essential to understand the severity of the problem and evaluate the potential impacts of management options in tackling soil erosion. Results showed that the average soil erosion rate in the study watershed was about 12 t ha^{-1} yr^{-1}. Though this rate of erosion is close to the tolerable limit, about half of the entire watershed experiences higher soil losses. When it comes to land use planning, quantitative accuracy is not as important as relative differences; it is the latter that is most helpful in prioritizing locations for intervention and in making decisions about management options.

We used a user-friendly modelling graphical interface to estimate soil erosion severity, map its spatial distribution and assess the impacts of different management practices. Mapping the spatial pattern of soil loss reveals that steeply sloping areas, gullies, and places with poor surface cover are characterized by comparatively high soil losses. Management efforts that targeted hotspots losing more than 10 t ha^{-1} yr^{-1} showed the most significant soil erosion reduction. Other options that targeted steep slopes (>25%) and gullies also showed significant soil erosion reduction. The modelling and simulation framework facilitated identifying site-specific and problem-oriented sustainable land management (SLM) and soil and water conservation (SWC) options that can reduce soil erosion risk and improve productivity.

In order to make interventions sustainable, it is essential to involve local communities in both identifying critical areas of erosion and in selecting suitable management options. Participatory approach can support farmers to reconsider and become more aware of the severity of soil erosion and the impacts of different management practices. The availability of an easy-to-use modelling framework can enhance the ability of local conservation leaders to use such tools in their land use planning with communities and facilitate ex-ante assessment of 'best-bet" SLM and SWC options.

Based on the outputs of the research, provisional recommendations can be made concerning the priority areas that should be targeted and the necessary SLM and SWC options required to tackle the observed soil erosion. In the future, more elaborate in-situ testing of the "promising approaches" identified in this study and suggested from other researchers need to be tested and demonstrated.

Detailed socio-economic and trade-off analyses are needed to inform best-bet and acceptable technologies vis-à-vis stakeholder preferences.

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Figure captions

Figure 1: Location of the study watershed in the Gudo Beret and Adisge Kebeles of the Amhara region, central highlands of Ethiopia.

Figure 2. Potential location of gullies derived based on Eq. (9) and (10) for parts of Gudo Beret and Adisge kebeles, Basona Woreda, central Ethiopia.

Figure 3. The spatial distribution of sediment delivery ratio as calculated based on Eq. (7).

Figure 4. Spatial patterns of net soil loss (t ha^{-1} yr^{-1}) experienced in the Gudo Beret-Adisge Kebeles of the Basonoa district, central Ethiopia.

Figure 5. Partial view of the different features of the modelling graphical interface to estimate soil loss and simulate impacts of management practices.

Figure 6. Scenarios of soil loss reduction for different land and water management practices in the Basona watershed, central Ethiopia. The bars show net soil loss rates estimated after conservation/management practices implemented targeting (from left to right): business as usual; steep slope areas of more than 20 degrees; gullies; gullies with 10 m buffer; gullies with 20 m buffer; steep slope + gullies; hotspot areas with soil loss more than 10 t ha^{-1} yr^{-1}; steep slope + hotspots; gullies and hotspots; and steep slopes, gullies and hotspots.

Figure 7. The spatial distribution of net soil loss considering (a) ‘business as usual’ practice and based on simulation of soil loss with different management options such as (b) areas of slope >20% conserved, (c) gullies treated, (d) areas with slope > 20% and gullies conserved; (e) hotspot areas of soil loss > 10 t $ha^{-1}yr^{-1}$ conserved, and (f) all options combined.

Figures

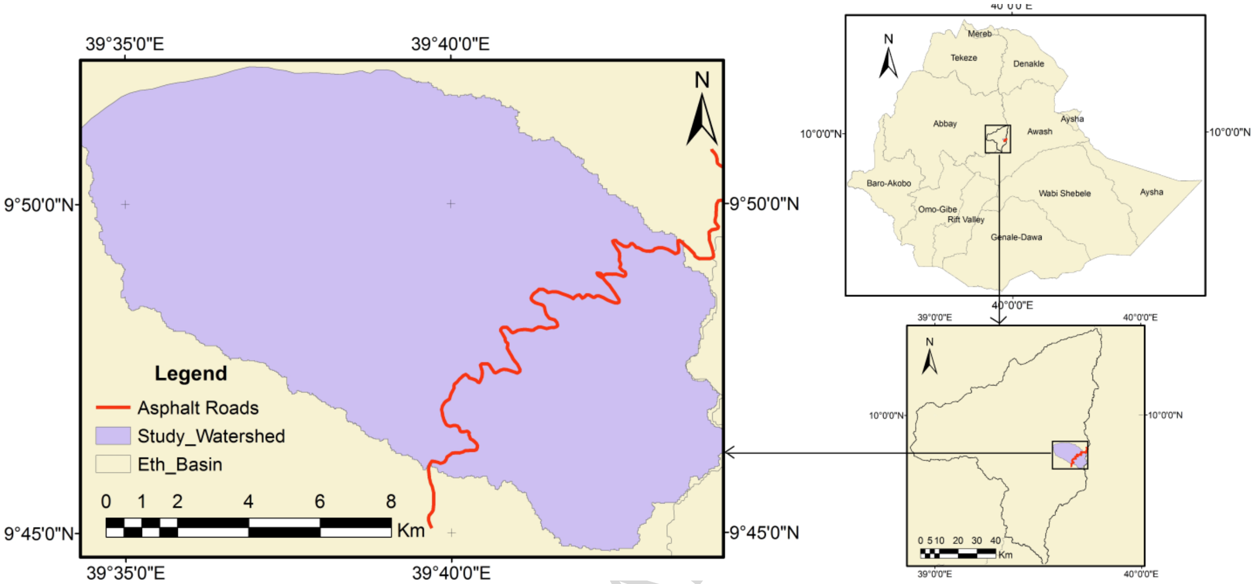


Figure 1

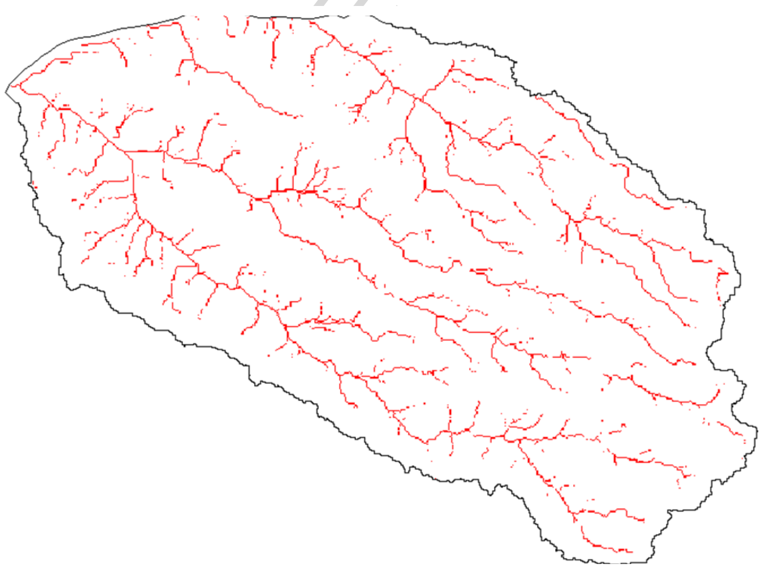
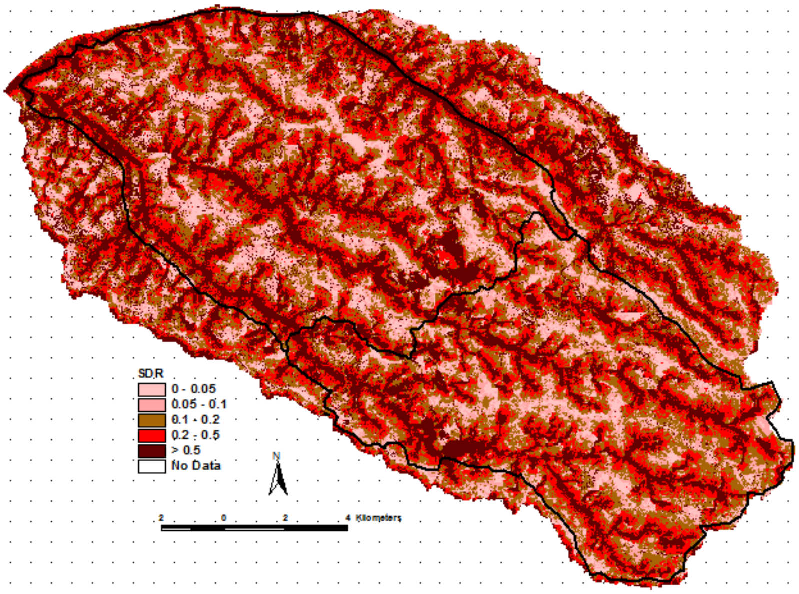


Figure 2



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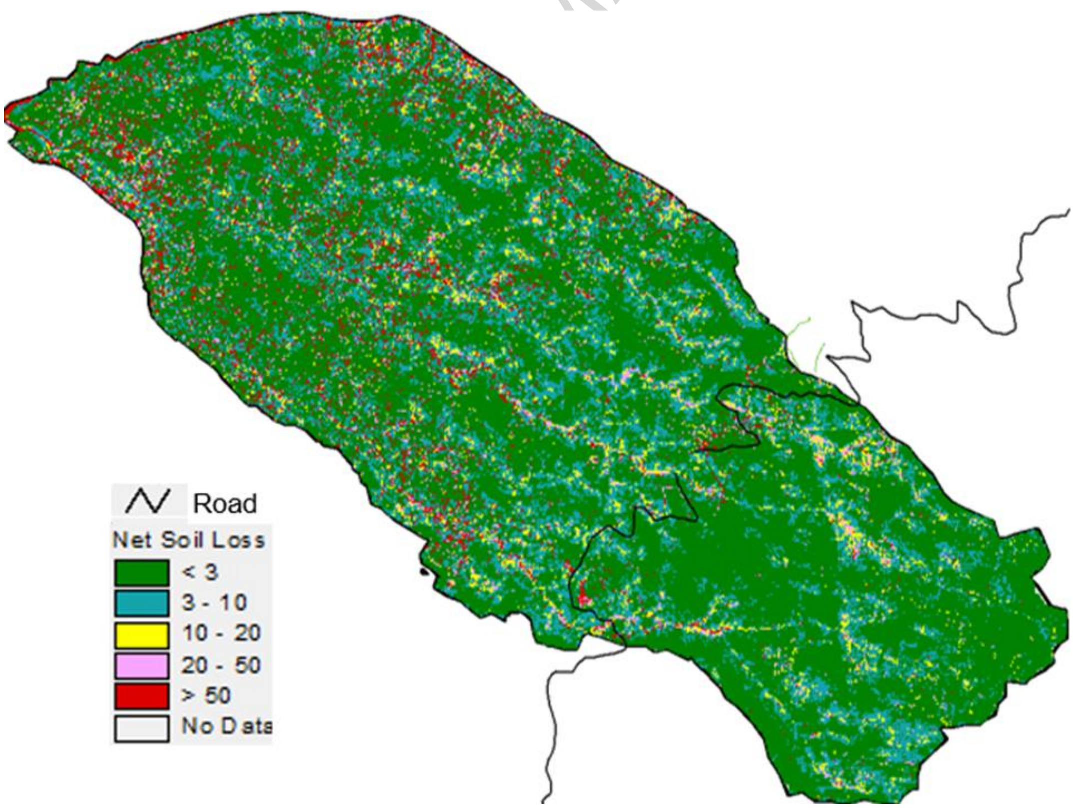

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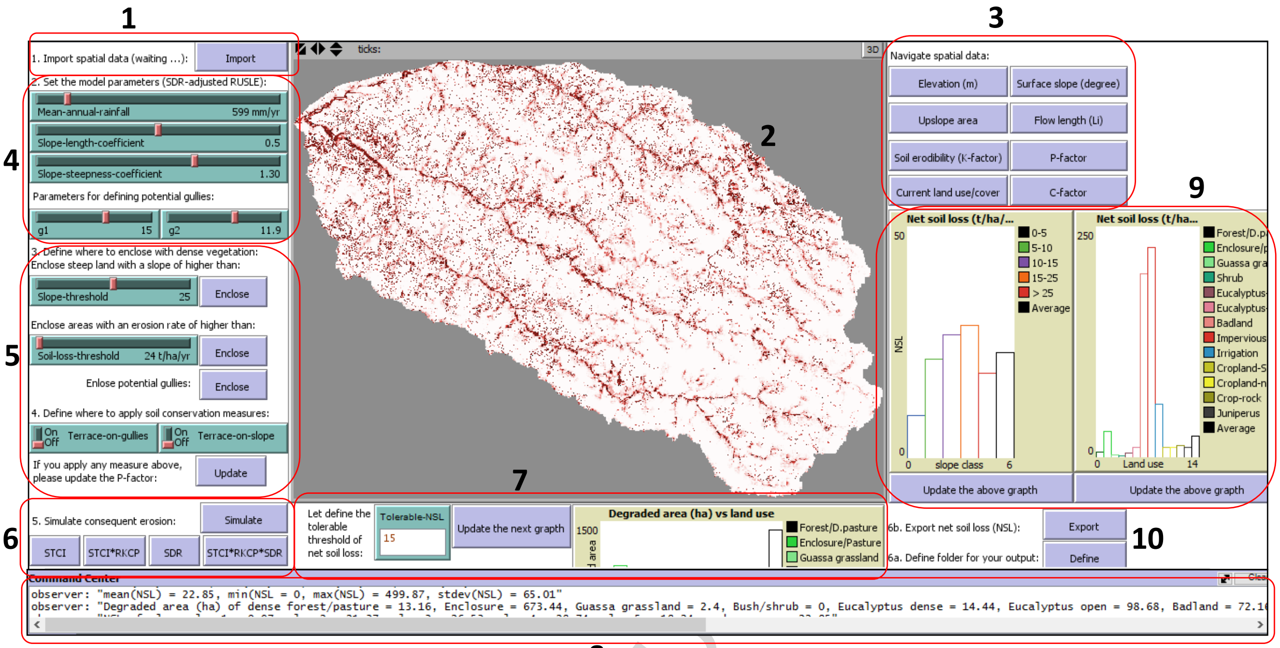

Figure 5

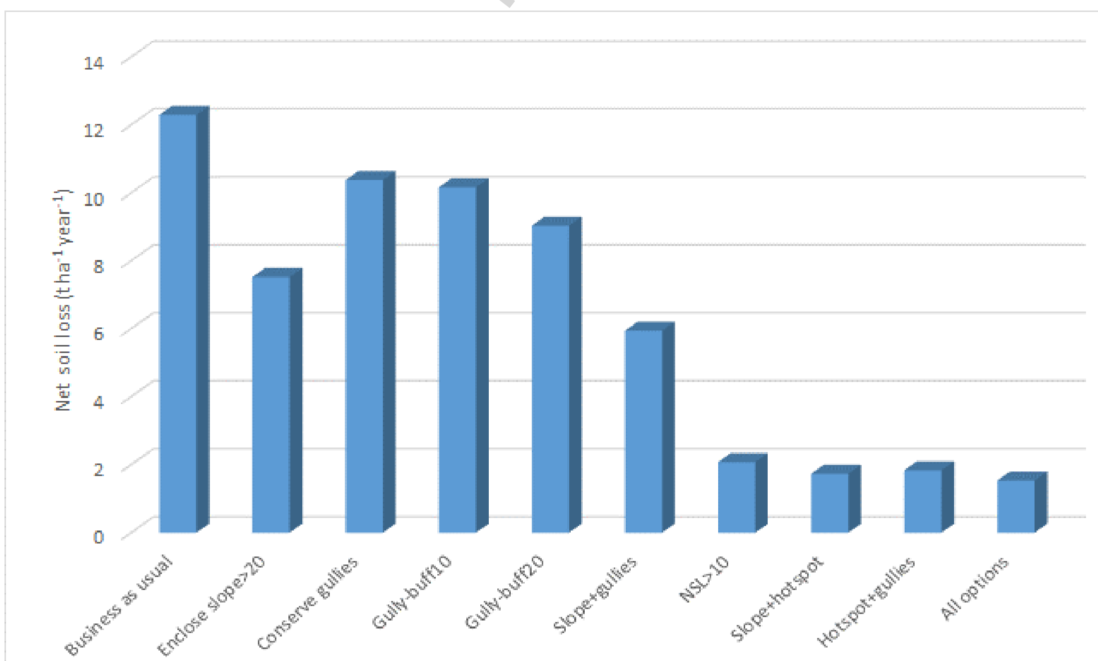

Figure 6

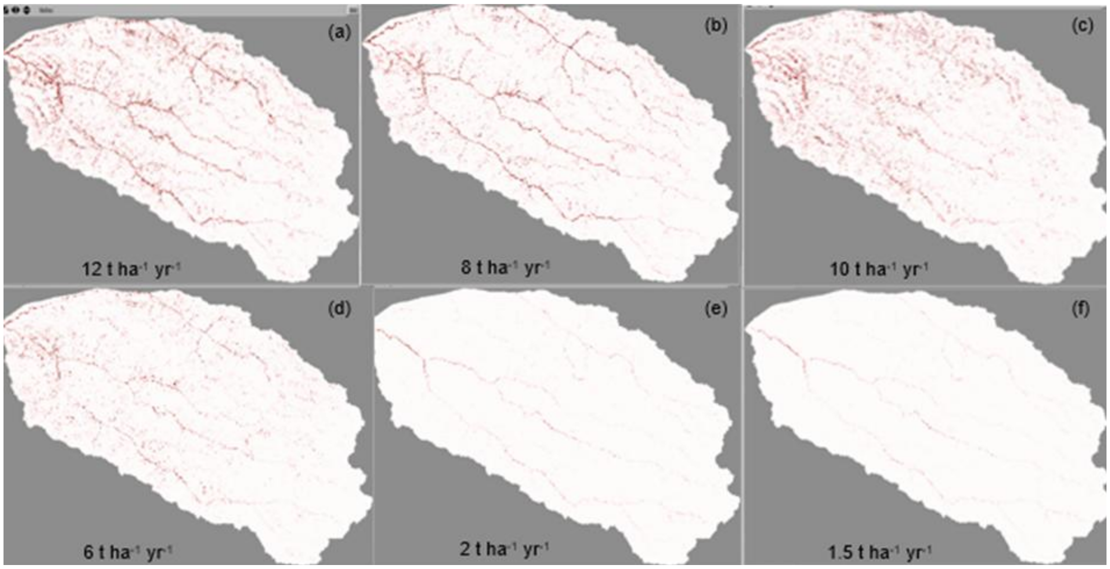
(a)
12 t ha-1 yr-1
(b)
8 t ha-1 yr-1
(c)
10 t ha-1 yr-1
(d)
6 t ha-1 yr-1
(e)
2 t ha-1 yr-1
(f)
1.5 t ha-1 yr-1

Figure 7

Tables

Table 1. Packing density – structure code associations (King et al., 1995)

Packing density (g cm^3)	Packing density class	Structure class	Structure code
< 1.4	Low	Good	1
1.4 – 1.75	Medium	Normal	2
> 1.75	High	Poor	3

Note: A "poor" class would be assigned to a soil exhibiting a massive or large ped structure while a "good" class would be found on highly developed soils with small peds (King et al., 1995).

Table 2. C-, and P-factors based on Hurni (1985); Machado et al. (1996); Eweg and Lammeren (1996) and surface roughness coefficient (Ri) based on Maidment et al. (1996); McCuen (1998); Stefano et al. (2005); Mutua et al. (2006) adapted for the study area

Land use/cover type	***C-factor***	***Management type***	***P-factor***	***Land cover description***	***Ri***
Dense forest	0.001	Ploughing up and down	1.0	Urban and built-up land	6.3398
Dense grass	0.01	Strip cultivation	0.8	Irrigated cropland and pasture	2.7737
Degraded grass	0.05	Stone cover (80 %)	0.5	Grassland	0.6401
Bush/shrub	0.02	Stone cover (40 %)	0.8	Dryland cropland and pasture	0.4572
Sorghum, maize	0.10	Protected areas	0.5	Shrubland	0.4572
Cereals, pulses	0.15	Ploughing on contour	0.9	Savanna	0.4267
Ethiopian Teff	0.25	Terraces	0.6	Cropland/Grassland mosaic	0.3962
Continuous fallow	1.00			Cropland/Woodland mosaic	0.3962

Mapping soil erosion hotspots and assessing the impacts of management practices using modelling and participatory approaches in the highlands of Ethiopia

Tamene et al.

Highlights

- Though quantitative information can be useful to understand the severity of soil loss, it is the spatial pattern that is more important for planning and targeting.
- Ex-ante analysis of the impacts of soil and water conservation measures in reducing soil loss is essential to inform researchers, planners and decision makers.
- An east-to-use tool to estimate the spatial dynamics of soil loss and conduct ex-ante analysis of the impacts of conservation measures is essential to design site- and context-specific options.
- Integrated management approaches applied at landscape scale can significantly reduce net soil loss and sustain overall productivity.
- Involving local communities in erosion risk assessment and evaluating impacts of conservation measures are essential to enhance technology adoption and out-scaling.