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Original Research Article

Assessing the soil erosion control efficiency of land management practices implemented through free community labor mobilization in Ethiopia

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ABSTRACT

This study aimed to assess the influence of conservation practices (P) and cover management (C) on soil loss reduction by determining it at the scale of landscape units in 16 systematically selected watersheds. Focusing on major land management practices implemented through free community labor mobilization, the assessment combined remote sensing techniques, field observation, and expert as well as local knowledge. The results show an average net decrement of 39% (± 19%) in the P factor value and 8.9% (± 21%) in the C factor value after implementation of land management practices. P factor value reduction is linked to a high area coverage of level structures, while increases in the P factor value are associated with poor quality of structures, inappropriate practices, and wide spacing between structures on steep slopes. C factor value reduction is observed in non-arable shrub- and bushland with enriched area closure, whereas increased C factor values are associated with open access grasslands and untreated croplands. The overall change in P and C factor values resulted in a 42% (± 28%) relative soil loss reduction. The demonstrated approach makes it possible to assess spatial and temporal dynamics in the P and C erosion factors and to estimate spatially disaggregated changes in the P and C factor values. This can help to improve parameterization of inputs for erosion modelling and to assess their relative soil loss effect. The approach provides valuable feedback on watershed planning processes and supports informed decisions regarding the appropriate selection of land management practices.

1. Introduction

In Ethiopia, land degradation in the form of soil erosion is considered a severe problem that results in low agricultural productivity (Lemenih, Olsson, & Karltun, 2004; Nyssen et al., 2004) due to its on-site effects, as well as siltation of reservoirs (Haregeweyn et al., 2006, 2017) and economic impacts (Hurni et al., 2015). Based on this awareness of the problem, since 1970, various land management practices have been implemented in the country (Adimassu, Langan, Johnston, Mekuria, & Amed, 2016). Depending on their success, land management projects were renewed over several phases (Haregeweyn, Berhe, Tsunekawa, Tsubo, & Meshesha, 2012). In the 1970s and 1980s, such projects largely took a top-down approach to implementation. Beginning in the 1990s, this was gradually replaced by a bottom-up approach. Building on positive experiences with previous watershed management approaches and interventions, bottom-up land management efforts have recently begun to be implemented through uncompensated free labor, a modality also known as Free Community Labor Mobilization (FCLM) (Haregeweyn et al., 2012). Irrespective of the implementation approach followed, the implemented land management practices have visibly transformed Ethiopia's highland landscape, where the majority of dwellers are smallholder subsistence farmers (Alemayehu et al., 2009; Haregeweyn et al., 2012). However, the implementation of land management practices in a given watershed does not automatically lead to a reduction in soil loss; on the contrary, in some cases, soil loss might even increase. Therefore, the question of the impact of such endeavors across Ethiopia's rainfed agricultural areas, particularly in terms of the soil erosion control efficiency of

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measures implemented through FCLM, has remained high on the research agenda since the very beginning.

Soil erosion assessments across large areas for which long-term observation data are lacking, are often done by means of models (Cuomo & Della Sala, 2015; Cuomo, Della Sala, & Novità, 2015). Generally, soil erosion models are categorized as physical and empirical, and their applicability varies depending on the input data required (Cuomo et al., 2015). There are globally used soil erosion assessment models that work with very general and comparably few input data and can be applied over large areas (Lee & Lee, 2006; Renschler, Mannaerts, & Diekkruger, 1999). The Universal Soil Loss Equation (USLE) has been applied to understand dominant erosion mechanisms by means of quantitative spatial analysis over large areas (Cuomo & Della Sala, 2015). In Ethiopia, the USLE has been used to assess the risk of soil erosion and the impact of land management practices on soil loss reduction at micro-catchment to basin levels (Gebremichael et al., 2005; Gelagay & Minale, 2016; Haregeweyn et al., 2017). Hurni et al. (2015) recently applied the Unit Stream Power-based Erosion and Deposition (USPED) model to assess the impact of soil and water conservation (SWC) in Ethiopia's rainfed agricultural area. The USLE and the Revised USLE (RUSLE) take a factor approach, determining the relative contribution of each erosion factor in the overall soil loss or soil reduction process (Angima, Stott, O'Neill, Ong, & Weesies, 2003; Blanco-Canqui & Lal, 2008; Renard, Foster, Weesies, McCool, & Yoder, 1997). The USLE contains six parameters, including the so-called conservation practices factor (P) and the cover management factor (C), which are largely modified by land management practices. Several authors of model-based soil erosion assessments have pointed out that a lack of reliable data on the model parameters affects the quality of modelling results (Haregeweyn et al., 2017). Indeed, obtaining reliable spatial information on the P and C factors is challenging, as they are dynamic in nature.

Besides model-based assessments, a considerable number of studies have been conducted at the plot level to determine the magnitude of soil erosion and related impacts of land management practices (Adimassu et al., 2016; Grunder, 1992; Herweg & Ludi, 1999; Hurni, 1985). Plot-level long-term observation data on the P factor for different SWC practices and the C factor for different land cover types in Ethiopia are available from the Soil Conservation Research Project (SCRP) (Herweg & Ludi, 1999; Kaltenrieder, Hurni, & Herweg, 2007). However, plot-level studies are of limited use in understanding the impact of land management practices at a wider scale. Therefore, researchers integrate plot-level parameters in their models to produce spatially explicit soil erosion risk maps over large areas. Modelling soil erosion in a spatially explicit way over large areas, researchers like Adimassu et al. (2016), Haregeweyn et al. (2017) and Hurni et al. (2015) have demonstrated how P and C factors values can be translated from the plot level to the basin, subnational, and national levels. Likewise, relative change in soil loss due to sheet and rill erosion can be evaluated by looking at changes in the P and C factors over a given period of watershed management (Munro et al., 2008; Panagos, Borrelli, Meusburger, & van der Zanden, 2015).

When data are scarce, model-based assessments over large areas commonly use a single value each for the P and C factors, without considering their spatial variability. However, evaluating the spatial and temporal variability of soil erosion risk and reduction of soil loss due to changes in the parameters – especially the P and C factors related to human activities on the land – is crucial (Renschler et al., 1999). When assigning P factor values to different erosion control measures, it is commonly assumed that the relevant structures are optimally designed and spaced. The fact that SWC structures might not meet the optimal design and construction requirements and might therefore have a lower efficiency is often disregarded. Munro et al. (2008) considered relative quality differences when assigning P factor values to stone and soil bunds on arable and non-arable land and revealed a considerable difference in actual soil loss reduction efficiency. It is thus important to consider the quality of SWC structures when estimating P factor values to assess the impact of SWC on soil loss reduction. Overall, the spatial and temporal variability of soil erosion processes makes it necessary, when assigning P and C factor values, to consider the effectiveness of SWC practices based on the layout and design of structures, the period of construction, land cover types, agroecological settings, and farming systems. This can be achieved by integrating plot-level data with the models. Therefore, further important research questions with regard to large-scale soil erosion assessment are: What type of data are needed to assess and estimate the P and C factors, and at what scale (extent and detail)? How are these data best collected? And what techniques can be used to estimate and assign their values over large areas where experimental data are unavailable?

In sum, the effectiveness of a soil erosion assessment is affected by the extent or scale it focuses on, the availability of required data, and the feasibility of collecting the required data at the required scale and quality. Accordingly, the present study was conceptualized based on the following research problems: (1) Plot-level assessments provide accurate data, but these only apply to specific areas and fail to represent complex socio-economic and biophysical settings. (2) Assessments at the basin, subnational, or national level are commonly model-based and therefore result in less detailed information, which is of limited use to inform land management strategies and land use policies for Ethiopia's highly fragmented smallholder system. (3) Field-scale assessments can help to capture important dynamics and detailed information but are expensive and thus applicable only to relatively small areas. To obtain fairly detailed information across a large area, an intermediary approach is needed that capitalizes on the advantages of all existing approaches. This study consisted of developing and applying such an approach by integrating existing model-based, plot-level, and field-level erosion assessment techniques.

The main goal of this study was to assess the landscape transformations induced by the implementation of major land management practices through FCLM and the impact of these practices on soil loss reduction. We focused on identifying and measuring major land management practices considering the situation before and after FCLM. Our first objective was to assess the type, coverage, and quality of implemented SWC and cover management practices. Second, we aimed to assess and estimate the change in P and C factor values induced by these land management practices. Our third objective was to appraise the impact of the identified land management practices implemented through the FCLM approach by measuring relative soil loss reduction. Finally, we highlight sustainability implications related to the soil loss reduction efficiency of the FCLM approach.

2. Study sites

The FCLM approach has been widely implemented in Ethiopia since 2009 (Haregeweyn et al., 2012). To assess its efficiency and impact, we systematically selected 16 case study watersheds within the Amhara and Tigray regions, where FCLM had been widely implemented. Selection criteria included representation of the *Kolla* (lowland), *Weyna Dega* (midland), and *Dega* (highland) agroecological belts (Hurni, 1998); the various landforms (plain, rolling, undulating, hills, mountain); the major land uses and land covers; and the various farming systems (sorghum, maize, cattle, teff, pulses, wheat, barley). The location and basic characteristics of the 16 study watersheds are presented in Fig. 1 and Table 1, respectively.

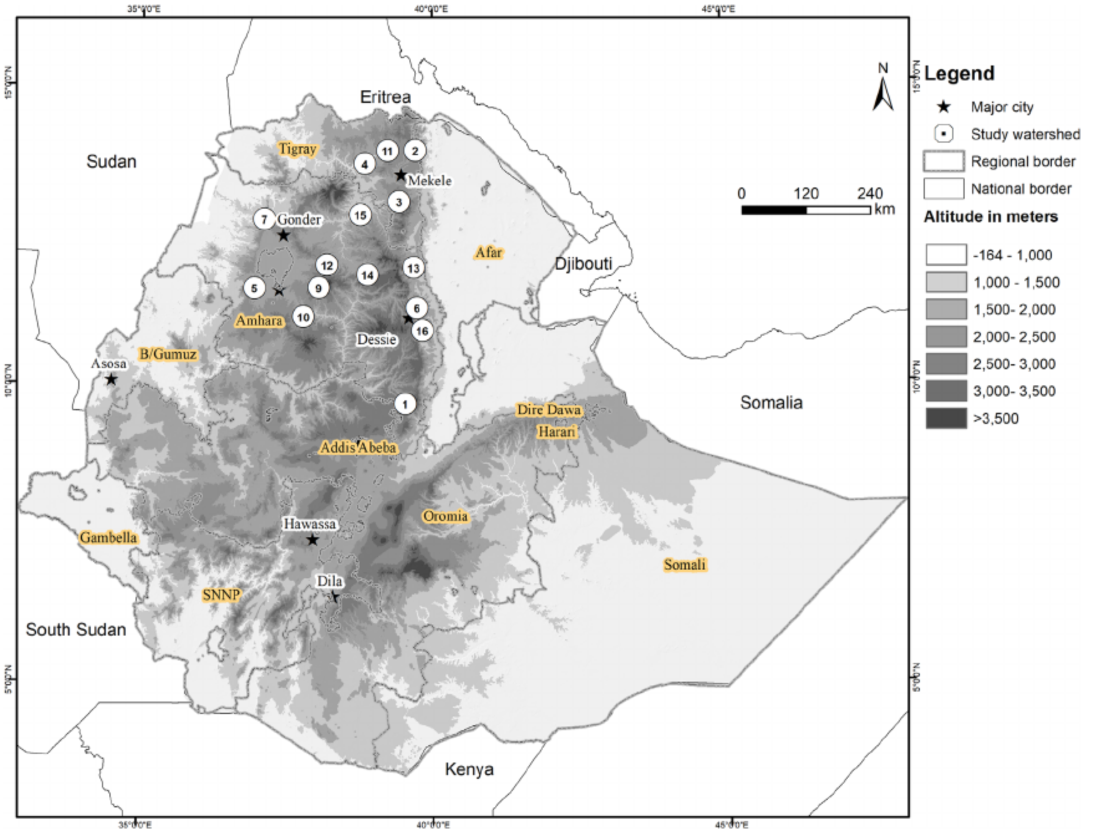

Fig. 1. Spatial distribution of the study watersheds.

Table 1
Location and characteristics of the study watersheds.

Code	Basin	Region	Woreda	Name of Watershed	X, Y coordinate at the outlet	Watershed area (ha)	Agro-ecology	Starting year
1	Abbay	Amhara	Angolela	Kemberie	39.55,9.57	284.9	Dega	2011
2	Tekeze	Tigray	Atsibi Womberta	Mehtsab Shimbra	39.73,13.83	465.6	Dega	2009
3	Tekeze	Tigray	Alaje	Telma	39.44,12.98	546.2	Dega	2005
4	Tekeze	Tigray	Kola Tembien	Beles	38,88,13.64	484.6	W/Dega	1998
5	Abbay	Amhara	South Achefer	Gelda	36.99,11.54	798	W/Dega	2011
6	Awash	Amhara	Tehulederie	Korebtit	39.76,11.20	479.7	W/Dega	2011
7	Tekeze	Amhara	Chilga	Lay Awga	37.15,12.67	860	W/Dega	2010
8	Tekeze	Tigray	Saisi Tsed Imba	Mariam Agamat	36.65,14.05	254.4	W/Dega	2008
9	Abbay	Amhara	East Istie	Mesal	38.09,11.59	410	W/Dega	2010
10	Abbay	Amhara	Hulet Iju Inesie	Teduma	37.81,11.05	296	W/Dega	2010
11	Tekeze	Tigray	Hawzen	Tonsoha	39.26,13.87	659.2	W/Dega	2009
12	Tekeze	Amhara	Farta	Wenjide	38.23,11.91	414	W/Dega	2011
13	Danakil	Amhara	Gubalafto	Amed Midir	39.71,11.89	366.5	Kolla	2011
14	Tekeze	Amhara	Meket	Gebriel	38.91,11.77	305	Kolla	2011
15	Tekeze	Amhara	Ziquala	Libam Sewir	38.81,12.78	325.7	Kolla	2008
16	Awash	Amhara	Dawa chefa	Timuga	39.85,10.81	287.6	Kolla	2011

3. Materials and methods

3.1. Data source and data collection techniques

To measure the induced landscape transformation, we considered two periods for analysis: before and after the implementation of the FCLM approach, corresponding to 2010 and 2015, respectively. Spatial information was generated from high-resolution satellite imagery available in Google Earth and Landsat images. A total of 16 Landsat images were used for the two periods of analysis. Detailed land use and land cover information for both periods was produced by integrating Google Earth images and Landsat images as described by Kassawmar, Eckert, Hurni, Zeleke, and Hurni (2016). Object-based feature extraction from high-resolution Google Earth images was the technique followed to map SWC structures as line features. For each period, we produced normalized difference vegetation index (NDVI) maps from the Landsat images using the Environment for Visualizing Images (ENVI) software. Further, we used the NDVI to measure the intensity of change in the vegetation cover.

The non-spatial information required to assess and estimate P and C factor values was collected by means of a semi-quantitative survey using a semi-structured questionnaire and field measurements. The information collected included types of crops grown,

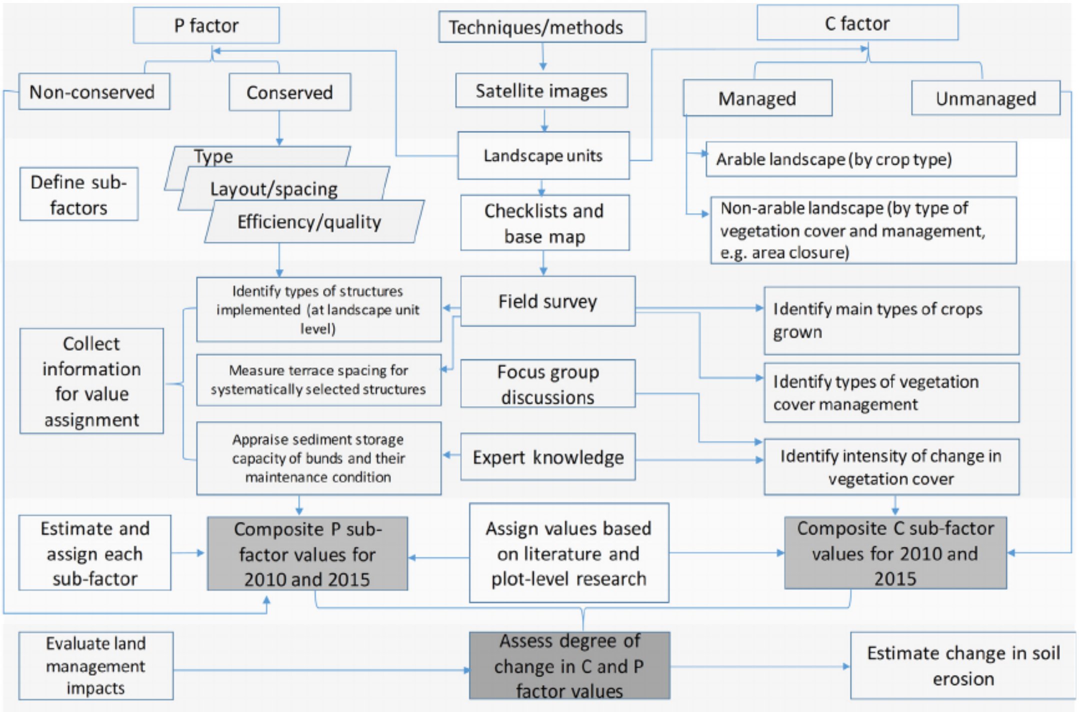

Fig. 2. Conceptual framework followed to assess and estimate P and C factor values.

land use changes and their intensity of change, as well as the type and quality of SWC structures implemented. These data were collected at the field scale, that is, within landscape units consisting of homogenous areas in terms of slope, land use and land cover, soil erosion, and type of land management practice. The landscape units were derived from high-resolution imagery available in Google Earth, following the procedures described in Kassawmar et al. (2016). The qualitative information collected based on experts' and local peoples' knowledge at the landscape unit level was then translated into numerical values. For that purpose, a comprehensive checklist with indicators was prepared based on a literature review. This integrated and participatory method of data collection and analysis enabled us to capture highly detailed information about land management practices, which is very difficult to do with conventional methods.

Fig. 2 illustrates the overall conceptual framework we followed to assess and estimate P and C factor values. It is unique in that it takes account of the following four elements of P and C factor assessment and assignment: First, it identifies and defines sub-factors of P and C specifically for the FCLM context. Second, it comprises assessment of the spatial extent and distribution of the sub-factors at the landscape unit scale and their share in the total area of a given watershed. Third, information from plot-level experiments is applied to estimate and assign sub-factor values; average P and C factor values at the watershed scale are then computed by weighting the sub-factor values according to their area share in the watershed. Fourth, temporal change of the sub-factor values is assessed and estimated by comparing the situation before and after implementation of the FCLM approach.

3.2. *Assessing and estimating the P factor*

3.2.1. *Description of the P factor*

The P factor reflects the influence of conservation practices on soil erosion. We assessed the P factor based on changes in supporting practices, which consist mainly in the implementation of physical SWC measures through FCLM. Several factors influence the efficiency of the implemented SWC structures, the main ones being the spacing, design, and stability of the structures (Subhatu et al., 2017). Achieving optimal and uniform design across watersheds is difficult in FCLM-based watershed management, where farmers' low technical capacity is frequently an issue. It is thus crucial to consider these influences on the P factor (Panagos, Borrelli, Meusburger, Alewell et al., 2015).

When data are scarce or study areas very large, modellers often use a single P factor value across large and complex landscapes, thereby only distinguishing conserved from non-conserved land (Hurni et al., 2015). The advantage is simplicity, as modellers only need to know a basic P factor value, which is commonly assigned based on the presence or absence of structures. In this study, based on the above considerations, we aimed at a more fine-grained assessment and assignment of P factor values that takes into account a number of sub-factors influencing them. In the context of FCLM, we found the type, layout, and stability of SWC structures to be the most important conditions affecting P factor values and soil erosion.

3.2.2. *The P_T sub-factor: spatial variation in type and coverage of the SWC structures implemented*

Often, in modelling soil erosion, a single P value is assigned for a watershed based on the presence and absence of SWC measures. However, the type of SWC measures implemented may vary considerably across the watershed. This spatial variation in type impacts overall erosion control efficiency at the watershed scale. Moreover, efficient implementation of land management practices requires proper selection of SWC measures considering the agroecological setting, topography, type of soil, and land use in a given watershed. In Ethiopia, SWC measures should, in principle, be selected in line with the recommendations given in the Community Based Participatory Watershed Development (CBPWD) Guideline (Desta, Carucci, & Wendem-Ageňehu, 2005). Nevertheless, in practice it may happen that the type of SWC measure selected is not the most appropriate one after all. Prior to the assignment of P factor values, we performed an exhaustive identification of the types of SWC measures present at the landscape unit level before and after the FCLM initiative. The basic P factor values for these SWC measures have generally been estimated based on long-term measurements of plot-level soil loss reduction. For example, the P factor value ranges are 0.20–0.37, 0.34–0.60, 0.12–0.25, 0.14–0.35, and 0.27–0.43 for graded fanya juu, graded bunds, level fanya juu, level bunds, and grass strips, respectively (SCRP, 2000). Similarly, in the review documents by Adimassu et al. (2016) and Grunder (1992), the average basic P factor values of 0.20, 0.60, 0.60, and 0.70 were assigned to level bunds, graded bunds, drainage ditches, and traditional bunds, respectively. Accordingly, for each landscape unit, the before and after P_T value was assigned for each of the identified SWC types. The P_T sub-factor in our study consists of the basic P factor, modified to reflect the degree of appropriateness of each identified type of SWC measure in the given landscape with reference to the CBPWD Guideline recommendations. For appropriately implemented types, values were assigned directly, based on the SCRP values. For inappropriately implemented types, a reduction factor was applied.

3.2.3. The P_S sub-factor: appropriateness of spacing between SWC structures

The CBPWD Guideline also contains recommendations on the optimal layout and spacing of structures for different landscape configurations and agro-climatic conditions. To reflect the extent to which the spacing of implemented structures is in line with these recommendations in a given landscape setting, we considered a sub-factor P_S. The value of this sub-factor was assessed by measuring the spacing between structures. For this purpose, we digitized structures from high-resolution Google Earth imagery for 2010 and 2015. Then, we calculated the average spacing between structures at the landscape unit scale based on the total length of structures in a given landscape unit (available from the structures shapefile) and the total area of that landscape unit. The area per unit length of structures was taken as the spacing between structures. This calculated spacing was verified by measuring the spacing of sample structures in representative landscape units in the field, and was then used to determine the P_S values. Spacing was considered appropriate if it was in line with the standard for the given slope class. For structures with appropriate spacing, P_S was assumed to be zero. For improperly spaced structures, P_S was estimated by determining the deviation of the calculated spacing from the spacing recommended for the given slope class in the CBPWD Guideline. However, the deviation in itself is not sufficient to assign P_S; in addition, it is necessary to know the relation between improperly and properly spaced structures. For this purpose, Hurni et al. (2015) developed an equation for the highlands of Ethiopia: $P = -0.0064 \times + 0.3$, where x represents the deviation of the actual spacing from the recommended spacing. The left term in the equation (i.e. $-0.0064 \times$) corresponds to the P_S sub-factor. Therefore, the deviation values were entered in this empirical equation to determine the P_S values. P_S is factored into the composite P factor value only if the actual spacing is greater than the recommended one; otherwise, P_S is equal to zero. To account for differences across the landscape units under consideration, we grouped the deviations into classes and calculated the median P_S value of each class (Table 2). The 1.5 m class interval was fixed after consulting the literature (Gessesse, 2009), according to which the soil loss reduction efficiency of SWC structures changes significantly when the spacing changes by 1.5 m or more.

3.2.4. The P_Q sub-factor: quality of SWC structures

To reflect spatiotemporal variation in the quality of SWC structures, we considered a P_Q sub-factor based on quality of construction (width and depth), stability of the structures, level of damage, and sediment retention capacity at the landscape unit level for 2010 and 2015. For structures of high quality, the P_Q value was assumed to be zero. For structures of medium to very low quality, a reduction factor value was estimated. Estimations were initially based on data from plot-level long-term observations on the efficiency of different SWC structures in retaining sediments (Grunder, 1992; Haregeweyn et al., 2012). These data were then combined with a qualitative rating of the design, stability, and proper integration of SWC measures in the landscape (Table 2). This rating was carried out by an interdisciplinary team of experts at the landscape unit level, based on a scale of 1–4 (1 = Very low quality, 2 = Low quality, 3 = Medium quality, and 4 = High quality).

3.2.5. Calculation of the composite P factor at the landscape unit and watershed levels

To obtain the composite P factor value at the landscape unit level, the three sub-factor values were added up as follows (Eq. (1)):

Table 2
Median P_S sub-factor values reflecting the deviation of actual from recommended spacing between SWC structures and P_Q sub-factor values reflecting the quality of SWC structures based on long-term plot-level observations and expert knowledge.

P_S sub-factor		P_Q sub-factor	
Class of deviation of structure spacing (recommended minus actual spacing)	Median P_S values ($P_S = -0.0064 \times$ and x = deviation of actual from recommended spacing)	Criteria for rating of SWC structure quality by experts	P_Q values based on plot-scale data and qualitative rating by experts
Dev $\geq$ 0	$P_0 = 0$	4 = High quality: Excellent design of structures, good stability, sufficient sediment retention capacity, integration with vegetative measures and runoff management structures (no need of maintenance in coming 2–3 years)	$P_Q = 0$
0 > Dev $\geq$ −1.5 m	$P_1 = 0.006$		
−1.5 > Dev $\geq$ −3.0 m	$P_2 = 0.015$		
−3.0 > Dev $\geq$ −4.7 m	$P_3 = 0.026$		
−4.7 > Dev $\geq$ −6.3 m	$P_4 = 0.035$		
−6.3 > Dev $\geq$ −7.8 m	$P_5 = 0.046$	3 = Medium quality: Satisfactory design of structures, moderate stability, moderate sediment retention capacity	$P_Q = 0.025$
−7.8 > Dev $\geq$ −9.3 m	$P_6 = 0.054$		
−9.3 > Dev $\geq$ −11.0 m	$P_7 = 0.066$	2 = Low quality: Poor design of structures, low stability, low sediment retention capacity (need for immediate maintenance)	$P_Q = 0.06$
−11.0 > Dev $\geq$ −12.5 m	$P_8 = 0.075$		
−12.5 > Dev $\geq$ −14.0 m	$P_9 = 0.085$		
−14.0 > Dev $\geq$ −15.7 m	$P_{10} = 0.096$	1 = Very low quality: Very poor design, no stability, no more sediment retention capacity	$P_Q = 0.13$
−15.7 > Dev $\geq$ −18.5 m	$P_{11} = 0.111$		
Dev < −18.5 m	$P_{12} = 0.126$		

$$P_{Composite} = P_T + P_S + P_Q \quad (1)$$

Where

$P_{composite}$ = Composite value of P factor
P_T = P factor assigned to different types of terraces
P_S = P increment factor due to sub-optimal spacing of terraces
P_Q = P increment factor due to sub-optimal quality of terraces

To obtain the composite P factor value at the watershed level, we first calculated the P_T, P_S, and P_Q sub-factor values for the watershed level. Each was calculated by weighting the landscape unit values according to the area shares of the different types of SWC structures, different spacing deviation classes, and different quality ratings, respectively. These weighted sub-factor values were then added up to obtain the composite P factor value at the watershed level. The same principle was applied for each period of analysis (2010 and 2015).

3.3. Assessing and estimating the C factor

3.3.1. Description of the C factor

The C factor reflects the influence of cover management on soil erosion. Its value is largely based on the type of vegetation cover and the extent to which this cover reduces the erosive power of rainfall and, thereby, sheet and rill erosion (Renard et al., 1997). We assessed the C factor based on vegetation cover changes on non-arable land and changes in the type of crops grown on arable land that happened after the implementation of watershed management.

3.3.2. Estimation of C factor values for arable land

Different types of crops have different C factor values (Panagos, Borrelli, Meusburger, Alewell et al., 2015). To properly estimate and assign C factor values in arable land, it is necessary to identify the crop types being grown. This was done by means of field observations, supported by focus group discussions. Crop type information was collected at the landscape unit level considering the situations before and after the implementation of watershed management. When a landscape unit contained two or more crop types, the dominant crop type was considered. However, the same crop type may have different C factor values depending on the agroecological conditions in the various study watersheds. A literature review was conducted to identify appropriate values for the crop types identified in each study watershed. The SCRP database provides detailed C factor values for individual crop types (Kaltenrieder et al., 2007). Hurni (1985) provides C factor values for different crop and land cover types in different agroecologial areas. In these sources, we found varying C values for the same crop in different agroecological areas. In order to consider these varying C values assignments for the same crop across our study watersheds, we calculated the average C factor values over the years. These average C values were assigned to each crop type identified at the landscape unit scale considering both periods of analysis (2010 and 2015).

3.3.3. Estimation of C factor values for non-arable landscapes

To assess and assign the C factor value for non-arable landscapes, information was collected on the types of vegetation cover as well as on land use and/or management considering the before and after situations. Using satellite imagery to map land cover requires ground truth information about the vegetation cover and the types of land management implemented for conservation, protection, enrichment, and other purposes. By means of a field survey and focus group discussions with experts and land users, such information was recorded for each landscape unit with regard to the situation before and after implementation of the measures. On this basis, we assigned C factor values within each landscape unit according to the estimations provided by Kaltenrieder et al. (2007) and Hurni (1985) for individual cover and land management types, for the situations before and after implementation of the measures.

3.3.4. Calculation of the aggregated C factor values at the watershed level

In order to compare the effectiveness of crop management and land cover management practices across study watersheds, we determined aggregated C factor values at the watershed level. To do that, the C factor values for arable and non-arable landscapes at landscape unit scale were weighted by their proportional area coverage in a given watershed and then added up to obtain a single C factor value for that watershed (Eq. (2)). The same principle was applied for each period of analysis (2010 and 2015).

$$C_{ws} = \sum_{1}^{n} Ci * \frac{Ai}{At} \quad (2)$$

Where

C_{ws} = Aggregated C value at watershed level
At = Total area of watershed
Ai = Area of landscape unit
Ci = C value at landscape unit level representing different types of land cover and land management

3.4. Analyzing the change in P and C values

As described above, changes in P and C factor values between 2010 and 2015 were assessed considering determinant sub-factors identified in each landscape unit of the study watersheds. When assessing the P factor, consideration was given to the type, spacing, and quality (design and stability) of the implemented SWC structures, as well as to the respective area coverage. When assessing the C factor, we considered crop types, vegetation covers, and land management, as well as their proportional area coverage. To assess the overall efficiency of the implemented land management practices, aggregated change in both P and C factor values was calculated for each watershed (Eqs. 3 and 4).

Change in P and C values was calculated as follows:

$$(P)_{change} = (Pcomposite)_{before} - (Pcomposite)_{after} \quad (3)$$

$$(C)_{change} = (C_{before} - C_{after}). \quad (4)$$

3.5. Appraisal of soil erosion control efficiency

The goal of this study was to assess the soil loss reduction efficiency of the land management practices implemented by means of FCLM. The assessment focused on estimation of the relative soil loss reduction due to change in P and C factors observed in the period between 2010 and 2015. The USLE factor approach is based on the relative contribution of each erosion factor to overall soil loss or soil loss reduction (Angima et al., 2003; Blanco-Canqui & Lal, 2008; Munro et al., 2008; Renard et al., 1997). In this study, we assumed all other factors involved in the USLE to remain unchanged over the considered period, and measured the relative contribution of only the P and C factors as done by Munro et al. (2008). This implies that change in soil loss due to sheet and rill erosion was evaluated exclusively based on change in the P and C factor values between 2010 and 2015. The evaluation was performed by multiplying the P and C values separately for the two periods (Eq. (5)).

$$\text{Percent soil loss reduction} = (\text{Soil loss before minus Soil loss after})/(\text{Soil loss before})*100$$
$$= \left[(P_{before}*C_{before})-(P_{after}*C_{after})\right]/(P_{before}*C_{before})\Big]*100 \quad (5)$$

4. Results and discussion

4.1. Assessment of landscape transformations based on the type and coverage of land management practices

In the study watersheds, the most commonly implemented land management practices affecting the P and C factors are: graded bund, level bund, traditional bund (including farm borders established across slope), ridge and furrow, area closure, enrichment, crop rotation, controlled grazing, and land use change (e.g. from unmanaged cropland to grassland or plantation). As noted by Haregeweyn et al. (2012), the implemented land management practices have substantially transformed the landscapes of the study watersheds and play a significant role in curbing land degradation in the country. We found that an average 68.5% (± 24.15%) of each study watershed was treated with SWC measures (Table 3). However, the proportional area coverage of SWC measures varied considerably across study watersheds. This variation is related to local communities' culture of practicing traditional terraces, periods of SWC programs, land use systems, and topography. This is supported by several other studies conducted in the region (Alemayehu et al., 2009; Gebresamuel, Singh, and Dick, 2010; Haregeweyn et al., 2012). Watersheds in the northwestern part of the study area have a relatively low coverage of SWC practices (30–65%) compared to those in the northeastern parts (over 85%). In many of the northwestern watersheds, SWC measures were not traditionally implemented, whereas communities in the northeastern watersheds already practiced construction of SWC structures before the inception of FCLM-based watershed management. Besides, several SWC campaigns had been implemented in these areas through Food-for-Work initiatives, although many of these structures have had to be rebuilt due to failure and design faults as noted in Haregeweyn et al. (2017) and Nyssen et al. (2004).

The result of a detailed land use and land cover change analysis performed at the landscape unit scale is summarized at the watershed level in Fig. 3 and Table 4. Such detailed analyses are missing in many similar model-based studies (Haregeweyn et al., 2017; Hurni et al., 2015). On average, 38.7% (± 22.8%) of each watershed shows a substantial or slight improvement in land use and land cover. Large areas of arable land remain arable, while significant areas of grassland were converted to cropland. Before the inception of FCLM-based watershed management, the majority of non-arable areas were freely grazed and appeared bare and degraded. Change in the vegetation cover was apparent especially in the northeastern part of the study area, where the campaign introduced a major change in land use, for example from open access to controlled use systems, as well as from free to controlled movement of livestock. This is supported by other studies (Birhane, Teketay, & Barklund, 2007; Gebrehiwot & A, 2014; Mekuria et al., 2007). In many of the northwestern watersheds, however, change in the vegetation cover is slight, as free grazing has not yet fully stopped. Overall, like land management experience, changes in the types and degrees of land use and land cover vary considerably across the study watersheds depending on agroecological settings, farming systems, topography, land use, and enforcement of bylaws. As a result, the efficiency of land management practices and the contribution of FCLM vary across watersheds.

The intensity of cover changes induced by the implemented land management practices varies across the study watersheds and the landscape units of a watershed. This makes it difficult to relate the degree of change with C value change for all watersheds and for all landscape units in each watershed. To estimate the efficiency of land management change, we established the following indicators adapted from Munro et al. (2008): no change, substantial improvement, slight improvement and deterioration (Table 4).

4.2. Efficiency of the implemented SWC structures

Assessing the efficiency of land management practices is difficult due to the lack of the required data at the required scale. For instance, Nyssen et al. (2004) examined the effectiveness of SWC measures by assessing soil bunds in a specific watershed. However, findings from such approaches cannot be generalized across large and complex areas. This study assessed the effectiveness of the FLCM-based watershed management approach across several representative watersheds based on two measures of efficiency of implemented SWC measures: quality of construction and layout/spacing. The former was assessed by means of a qualitative rating

Table 3
Area coverage of different types of SWC practices by watershed (in percent).

Watershed code	Area (ha)	Area coverage (%) per type of SWC practice				Total SWC coverage (%)	Area without SWC (%)
		Graded bund	Level Bund	Traditional Bund	Ridge and furrow		
1	284.9	1.49	5.16		71.92	78.58	21.42
2	465.6		97.00			97.01	2.99
3	546.2		11.12	75.70		86.40	13.28
4	484.6		95.95			95.39	4.64
5	798.0	25.70	4.64			30.36	69.60
6	479.7	0.42	72.07		23.99	93.19	3.54
7	860.0		66.46			66.56	33.39
8	254.4		90.12			88.87	9.90
9	410.0	7.38	24.22			31.59	68.35
10	296.0		46.01			46.07	53.85
11	659.2		33.78			33.78	66.22
12	414.0		48.95			49.86	51.03
13	366.5		63.75			63.76	36.24
14	305.0	0.24	62.29			62.64	37.30
15	325.7		96.27			96.27	3.73
16	287.6		76.16			76.16	23.85
Mean		**7.04**	**55.87**	**75.7**	**47.95**	**68.52**	**31.45**
Standard dev.		**10.83**	**32.48**			**24.15**	**24.50**

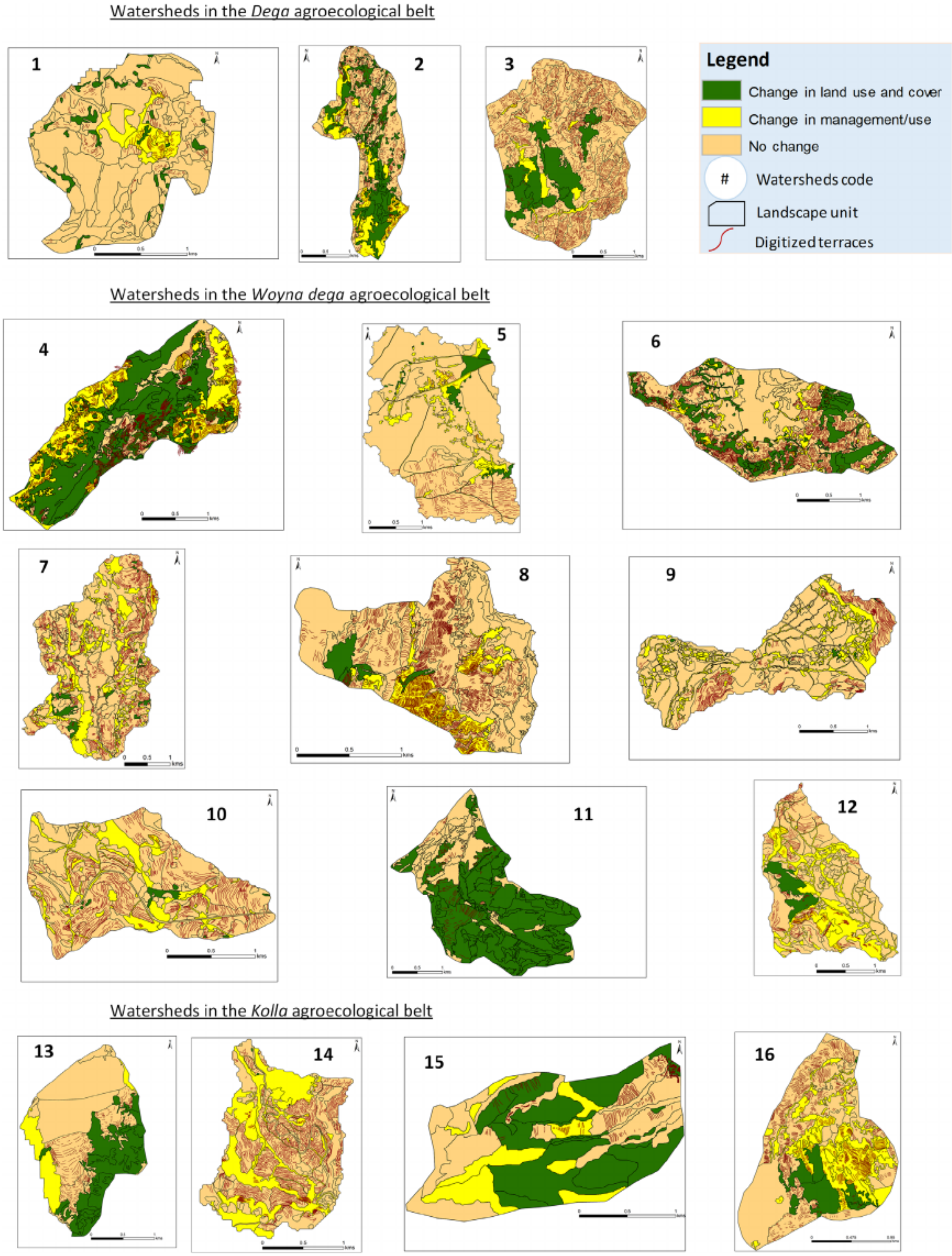

Fig. 3. Land use and land cover changes and coverage of terraces following the implementation of watershed management in the study watersheds.

of physical SWC structures, while the latter was evaluated based on the deviation of actual from recommended structure spacing (see Table 2). Results show that the majority of implemented SWC measures are of low to very low quality (Table 5). Such poor efficiency is widely observed in the highlands of Ethiopia and is mainly due to inappropriate layout and a lack of regular maintenance (Subhatu et al., 2017). Quality was generally higher in the northeastern watersheds, as communities there have many years of experience in constructing and maintaining terraces.

Table 5 presents the results of the assessment of structure spacing for each slope class. Positive values indicate unnecessarily narrow spacing, whereas negative values indicate inappropriately wide spacing. The absolute deviation at different slope gradients ranges from 1 to 20 m. This indicates that deviations from optimal structure spacing contributed to the low performance of the majority of measures implemented through FCLM.

Table 4
Area coverage of land use and land cover change by watershed (in percent).

Watershed code	Area (ha)	Area showing no change (%)	Area showing change (%)			
			Substantial improvement	Slight improvement	Deterioration	Total area showing change
1	284.9	86.84	4.93	8.22	0.06	13.21
2	465.6	48.59	31.12	20.40	0.01	51.52
3	546.2	77.90	17.83	4.27	0.01	22.11
4	484.6	13.36	55.37	31.27	0.02	86.65
5	798.0	86.73	3.71	9.56	0.05	13.32
6	479.7	39.30	56.97	3.73	0.01	60.71
7	860.0	75.95	2.69	21.36	0.04	24.10
8	254.4	79.02	4.74	16.23	0.32	21.30
9	410.0	82.13	0.42	17.44	0.05	17.92
10	296.0	82.36	1.39	16.25	0.27	17.91
11	659.2	24.63	75.37	0.00	0.00	75.37
12	414.0	62.48	7.20	30.32	0.06	37.58
13	366.5	59.49	30.33	10.18	0.03	40.54
14	305.0	66.28	0.05	33.67	0.03	33.75
15	325.7	38.09	44.90	17.01	0.01	61.93
16	287.6	58.64	12.25	29.12	0.01	41.38
Mean		**57.75**	**21.83**	**16.81**	**0.06**	**38.71**
Standard dev.		**22.87**	**24.31**	**10.47**	**0.09**	**22.83**

Table 5
Proportional area coverage of SWC structure quality classes in percent of the total area covered by SWC measures, and average deviation in meters of actual from recommended structure spacing.

Watershed code	Shares of quality ratings in overall area covered by SWC structures (%)				Average deviation of actual from recommended structure spacing (m) per slope class						
	Very low	Low	Medium	High	0–3%	3–8%	8–15%	15–30%	30–50%	> 50%	Mean
1		53.66	46.34				−2.37	−5.53			−3.95
2	1.06	37.91	52.04	8.99	−1.00	5.19	−4.14	−7.61	−19.20		−5.35
3		90.16	8.82	1.01	4.61			−8.68	−9.92		−4.66
4	1.19	29.25	67.72	1.85		2.90	−5.80	−4.35	−7.35	−4.50	−3.82
5		66.01	6.05	27.94		−2.86					−2.86
6		23.10	57.63	19.27		8.73	0.79	−5.36	−6.53	−5.00	−1.47
7	19.44	13.08	44.03	23.45		−0.58	0.22	0.72	2.26		0.65
8		53.82	46.18				−2.97	−2.12	−16.75		−7.28
9		65.70	34.30				5.13	1.87			3.50
10	19.01	17.73	38.75	24.50		8.75	−2.29	4.05			3.50
11		50.00	4.12	45.88		−1.97	−2.71	1.63			−1.02
12	39.16	23.67	37.17			−12.1	7.95	4.28	3.56		0.92
13	22.01		62.88	15.11			1.48	−1.27			0.11
14	1.29	1.48	9.05	88.17			0.96	4.95	3.19		3.03
15	11.47	42.12	46.41			13.48	2.21	−8.00			2.56
16		71.33	28.35				0.36	−0.21	−3.50		−1.12
Mean	**14.33**	**42.60**	**36.86**	**25.62**							

4.3. Estimated change in P and C factor values

4.3.1. Change in P factor values

Many previous SWC impact assessment studies (Haregeweyn et al., 2017; Hurni et al., 2015) missed the effect of the decline over time of land management practices' effectiveness in controlling soil erosion. Many studies consider only one point in time and base their assessment on the presence or absence of land management practices. This study additionally considered the fact that P factor values change with time and space. Table 6 presents estimated P factor values before and after implementation of the measures and change in P factor values for each watershed. Before the implementation of FCLM, the P factor value was generally higher ($\sim$ 1.0), as most watersheds had no SWC measures at all. The high P factor values ($>$ 0.6) after implementation observed in some watersheds are due to wide spacing and inappropriate choice of structures (e.g. traditional bunds and excessive drainage ditches). Watersheds where SWC had been practiced before implementation of the FCLM approach (on cultivated and degraded land) were assigned an initial P factor value of 0.7–0.9. The average P factor value in these watersheds was 0.54 ± 0.21 after implementation, compared to 0.88 ± 0.12 before implementation. No watershed had a P factor value of less than 0.25 after implementation, as all of them had issues with quality and spacing. Overall, our analysis of SWC coverage indicates that the FCLM campaign achieved a reduction in P factor values by about 39% (8–75%). In 9 out of 16 study watersheds, the P factor value was reduced by at least 0.35. Three watersheds show an estimated reduction of less than 0.1, due to large shares of traditional or inappropriate bunds, low area coverage of SWC practices, and wide structure spacing.

The average weighted P factor values for the study watersheds after implementation were found to range between 0.25 and 0.92. As presented in Table 7, watershed-level P factor values after implementation increased by 0.00–0.057 due to poor spacing of structures and by 0.005–0.074 due to poor quality. The highest implementation quality was achieved in the Gebriel watershed (No. 14; increase by only 0.009), whereas the poorest implementation quality was found in the Wenjidie (No. 12) and Telma (No. 3) watersheds (increase by 0.077, mainly due to poor

Table 6
P and C factor values before and after the implementation of FCLM, and their change.

Watershed code	P factor				C factor			
	P_before	P_after	Change in P	% change in P factor	C_before	C_after	Change in C	% change in C factor
1	1.00	0.735	0.265	26.45	0.308	0.361	−0.054	−17.43
2	0.70	0.294	0.406	58.00	0.248	0.212	0.036	14.53
3	0.90	0.801	0.099	10.95	0.355	0.372	−0.016	−4.57
4	0.70	0.302	0.398	56.88	0.417	0.320	0.097	23.19
5	1.00	0.918	0.082	8.18	0.512	0.594	−0.082	−15.94
6	0.70	0.369	0.331	47.28	0.487	0.477	0.010	2.06
7	1.00	0.530	0.470	46.95	0.563	0.548	0.015	2.68
8	0.70	0.336	0.364	52.07	0.443	0.489	−0.046	−10.29
9	0.90	0.828	0.072	8.02	0.283	0.331	−0.048	−17.05
10	0.90	0.680	0.220	24.46	0.342	0.314	0.029	8.43
11	0.90	0.560	0.340	37.80	0.431	0.353	0.078	18.15
12	1.00	0.647	0.353	35.28	0.363	0.301	0.062	17.03
13	1.00	0.526	0.474	47.45	0.133	0.054	0.078	58.94
14	0.80	0.448	0.352	43.99	0.179	0.157	0.022	12.34
15	1.00	0.251	0.749	74.87	0.160	0.092	0.068	42.33
16	0.90	0.446	0.454	50.48	0.563	0.519	0.045	7.95
Mean	**0.88**	**0.54**	**0.34**	**39.32**	**0.36**	**0.34**	**0.02**	**8.90**
Standard deviation	**0.12**	**0.21**	**0.17**	**19.28**	**0.14**	**0.16**	**0.05**	**21.04**

Table 7
Weighted P sub-factor values due to structure spacing (P_S) and quality (P_Q) at the watershed level.

Watershed Code	Weighted P_S	Weighted P_Q	Total weighted P reduction ($P_S + P_Q$)
1	0.029	0.044	0.073
2	0.033	0.037	0.070
3	0.057	0.056	0.113
4	0.028	0.036	0.064
5	0.024	0.041	0.065
6	0.027	0.028	0.055
7	0.019	0.044	0.063
8	0.024	0.044	0.068
9	0.003	0.048	0.051
10	0.005	0.045	0.050
11	0.025	0.031	0.056
12	0.003	0.074	0.077
13	0.000	0.044	0.044
14	0.004	0.005	0.009
15	0.007	0.052	0.059
16	0.005	0.050	0.055
Mean	**0.018**	**0.042**	**0.0609**
Standard deviation	**0.015**	**0.015**	**0.0210**

structure quality, and by 0.114, mainly due to wide structure spacing, respectively). The rest of the watersheds showed increases in P factor values by 0.03–0.05 due to poor structure quality and by 0.01–0.035 due to wide structure spacing. This means that, regardless of any other erosion factors, poor implementation quality causes an average increase in soil loss by 5–10% at the watershed scale (250–850 ha).

4.3.2. Change in C factor values

Although the magnitude of change in C factor values was not as high as that in P factor values (0.34 ± 0.16 on average), the average change nonetheless amounted to 0.02. Changes in land cover were observed in all watersheds (Table 6), but they varied in intensity and direction. As shown in Table 6, an increase in the C factor value was observed in 11 watersheds and a decrease in 5 watersheds. However, as indicated in Table 4, the landscape unit level assessment shows that in 58% of the total area of all study watersheds, the C factor value remained unchanged. The variations in C factor values across watersheds and the magnitude of their changes are largely determined by the extents of arable and non-arable land before implementation, the spatial configuration of land uses in the topo-sequence, and the agroecological settings. The lowest weighted average C factor values after the implementation of watershed management were identified in watersheds with a large share of well-managed shrub- and bushland (No. 13 and 15). The highest weighted average C factor values were found in watersheds where cereal crop farming and extensive grazing are the dominant land use systems (No. 5, 7, and 16). Positive changes in weighted C factor values are prominent where degraded communal land has been converted to managed area closures and woodlots. Prohibition of open grazing, which is a key element of land management in Ethiopia, has contributed greatly to improved C factor values. This is reflected in the C factor value changes achieved where land cover and land use types were changed as a priority action in FCLM.

An increase in C factor values was observed in watersheds situated in the *Dega* and *Weyna Dega* agroecological zones. These watersheds were predominantly covered by cropland both before and after the implementation of FCLM, so a deterioration in the vegetation cover of non-arable lands was responsible for the increase in C factor values. On the other hand, we found substantial decreases in C factor values in watersheds located in the *Kolla* agroecological zone, where the temperature favors improved vegetation cover, and moderate decreases in some watersheds in the *Weyna Dega* agroecological zone. More importantly, in the *Kolla* zone, the presence of non-arable landscapes mainly on steep slopes enables the implementation of extensive vegetation improvement practices such as area closure and enrichment plantation. For that reason, vegetation improvement and C value decreases are most prominent in watersheds in the *Kolla* zone.

4.4. Aggregated changes in P and C factor values and their implications for soil loss reduction

Aggregated changes in P and C factor values and the relative soil loss reduction rates are summarized in Table 8. The aggregated change in P and C factor values was about 0.12. Consequently, we estimate the combined P and C factor effect on soil loss through sheet and rill erosion in 2015, after the implementation of FCLM, to have amounted to only 58% of what it was in 2010, before implementation. Assuming that the other USLE factors remained constant over the considered period of analysis, as implemented in Munro et al. (2008), the change in P and C factor values resulted in

Table 8
Aggregated change in P and C factors and its effect on soil loss reduction.

Watershed code	PC_before	PC_after	Change in PC	PC_after as percent of PC_before	Relative percentage of soil loss reduction
1	0.308	0.266	0.042	86.37	13.63
2	0.174	0.062	0.112	35.90	64.10
3	0.320	0.298	0.022	93.12	6.88
4	0.292	0.097	0.195	33.12	66.88
5	0.512	0.545	−0.033	106.46	− 6.46
6	0.341	0.176	0.165	51.63	48.37
7	0.563	0.291	0.272	51.63	48.37
8	0.310	0.164	0.146	52.86	47.14
9	0.254	0.274	−0.02	107.66	−7.66
10	0.308	0.213	0.095	69.17	30.83
11	0.388	0.197	0.191	50.91	49.09
12	0.363	0.195	0.168	53.70	46.30
13	0.133	0.029	0.104	21.58	78.42
14	0.144	0.071	0.073	49.10	50.90
15	0.160	0.023	0.137	14.49	85.51
16	0.507	0.231	0.276	45.58	54.42
Mean	**0.32**	**0.20**	**0.122**	**57.71**	**42.29**
Standard dev.	**0.13**	**0.13**	**0.09**	**27.94**	**27.94**

a 42% (± 28%) estimated relative soil loss reduction compared to the situation before implementation. The lowest soil loss reduction rates were found in study watersheds with a low area coverage and poor quality of SWC measures. A high soil loss reduction rate (> 65%) was found in four study watersheds with large changes in both P and C factor values. Relative soil loss was estimated to have been reduced by about 30–60% in many watersheds due to the expansion of SWC practices. In the two highland watersheds of Telma and Kemebrie (No. 3 and 1), the estimated soil loss reduction is very low, which is mainly because of the poor quality of SWC implementation. Overall, estimated soil loss reduction is higher in watersheds located in the *Kolla* agroecological zone and in some of the watersheds in the Weyna Dega zone; this is mainly owed to improved land use systems and vegetation management practices as well as the implementation of level bunds for soil moisture conservation. Overall, the estimated soil loss reduction rates at the watershed level range between 7% and 86% (± 28%). Two watersheds, however, show a 6–7% increase in soil loss compared to estimated soil loss before the implementation of watershed management. Considering all 16 watersheds and an average area coverage of physical SWC structures per watershed of 68%, the estimated average relative reduction in soil loss due to sheet and rill erosion attributed to the change in P and C factor values alone amounts to 42%. Assuming that the erosivity and erodibility factors remained unchanged over the period of analysis but considering change in the slope length factor in addition to the P and C factors, we would most probably find a further reduction of actual soil loss. Research conducted in the upper Blue Nile basin (Haregeweyn et al., 2017) and across Ethiopia's rainfed cropland (K. Hurni et al., 2015) found actual average soil loss reductions by 52% and 43%, respectively, on land treated with appropriate SWC measures. A review by Adimassu et al. (2016) shows that different SWC measures achieve soil loss reduction rates between 38% and 88%. Our results agree with these overall findings. Variations can arise due to the techniques applied, data quality, and the study area extent.

The relative soil loss reduction can be categorized into low, moderate, and high. Low soil loss reduction rates (− 6–15%) were estimated for only four of the 16 study watersheds; they are due to low positive changes in the P factor values (< 25% change) and high increases in C factor values (negative percent change). The four watersheds are located in the *Dega* and *Weyna Dega* agroecological zones and experienced no land cover change and no area closure interventions. Moderate soil loss reduction rates (30–65%) were estimated for ten watersheds and are primarily due to decrement in the P factor values (25–60% change) and low positive changes in C factor values (0–20% change). Most watersheds with a moderate estimated soil loss reduction are found in the *Weyna Dega* agroecological zone; two are in the *Kolla* zone but showed low C factor change, and one is in the *Dega* zone. High soil loss reduction rates (80–85%) were estimated for only two of the 16 study watersheds; both are located in the *Kolla* zone and showed considerable decrement in both the P (> 50% change) and C (40–60% change) factors value.

5. Conclusion

This study aimed at assessing the landscape transformation induced by the implementation of major land management practices. Given that many of the land management practices are implemented at smallholder farmers' plots level (< 0.25 ha), it was difficult to apply existing assessment procedures. One of several major challenges in assessing the efficiency of the implemented land management practices was to identify and estimate the factors contributing to erosion at a spatial and temporal scale that matches the coverage and duration of the FCLM program. To address these challenges, we developed an approach that capitalizes on the strengths of existing approaches while overcoming applicability limitations in complex socio-economic conditions and diverse agroecological settings. The approach is innovative in that it considers spatiotemporal variations in erosion sub-factors and assigns them at a landscape unit scale, which is of paramount importance for soil erosion assessment in heterogeneous smallholder farming areas. Applying this approach, our study has demonstrated how relevant, comprehensive, complete, and accurate information on any land management approach can be collected and used to assess the efficiency of the implementation approach. However, this study focused only on two erosion factors, P and C, which were directly related to the studied FCLM program, vary in time and space, and affect erosion processes in short periods of time. Such an assessment approach does not show the absolute efficiency of land management practices; in fact, it may underestimate the efficiency of the FCLM program. A more accurate impact assessment could be achieved by considering all other integrated sustainable land management practices, including the different soil fertility management techniques. Indeed, the consideration of many factors is resource (time and finance) demanding and makes such comprehensive assessment techniques difficult to apply across large basins. For this reason, we applied our approach to 16 representative case study watersheds so as to help extrapolate local-scale findings to a larger basin or region. In sum, our assessment approach is capable of providing timely feedback regarding strengths and weaknesses of the FCLM program and is thus useful for supporting sustainable land management at the national level.

The assessment revealed that the implementation of sustainable land management practices has substantially transformed Ethiopia's smallholder farming landscape. About 70% of the landscape of the study watersheds is covered by physical SWC measures. Our findings also show that change in P factor values is greater than change in C factor values. However, the changes in P and C factor values vary considerably across the study watersheds. This may be attributed to variation in the types and coverage of land management practices, as well as to differences in the quality of implementation based on the experience, skill, and commitment of the implementing experts and communities. Based on our findings, we conclude that although the FCLM approach has achieved substantial area coverage of physical SWC measures, more attention should be paid to the appropriate selection, design, and layout of SWC measures. Moreover, specialized and efficient land management practices should be implemented selectively in specific agroecological zones—for example, land use changes such as area closure in the *Kolla* zone. Finally, we recommend that the approach should include monitoring and evaluation as part of the process in order to ensure regular maintenance of physical SWC measures and enforcement of local bylaws.

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References

Adimassu, Z., Langan, S., Johnston, R., Mekuria, W., & Amed, T. (2016). Impacts of soil and water conservation practices on crop yield, run-off, soil loss and nutrient loss in Ethiopia: Review and synthesis. *Environmental Management*.

Alemayehu, F., Taha, N., Nyssen, J., Girma, A., Zenebe, A., Behailu, M., Deckers, S., et al. (2009). The impacts of watershed management on land use and land cover dynamics in Eastern Tigray (Ethiopia). *Resources Conservation and Recycling*, *53*(*4*), 192–198.

Angima, S. D., Stott, D. E., O'Neill, M. K., Ong, C. K., & Weesies, G. A. (2003). Soil erosion prediction using RUSLE for central Kenyan highland conditions. *Agriculture Ecosystems and Environment*, *97*(*1–3*), 295–308.

Birhane, E., Teketay, D., & Barklund, P. (2007). East African journal of sciences (2007) enclosures to enhance woody species diversity in the dry lands of Eastern Tigray. *Diversity*, *1*, 136–147.

Blanco-Canqui, H., & Lal, R. (2008). *Principles of Soil Conservation and Management. Uma ética para quantos?* (Vol. XXXIII). Retrieved from ⟨http://www.americanbanker.com/issues/179_124/which-city-is-the-next-big-fintech-hub-new-york-stakes-its-claim-1068345-1.html%5Cnhttp://www.ncbi.nlm.nih.gov/pubmed/15003161%5Cnhttp://cid.oxfordjournals.org/lookup/doi/10.1093/cid/cir991%5Cnhttp://www.scielo.cl/pd⟩.

Cuomo, S., & Della Sala, M. (2015). Large-area analysis of soil erosion and landslides induced by rainfall: A case of unsaturated shallow deposits. *Journal of Mountain Science*, *12*(*4*), 783–796.

Cuomo, S., Della Sala, M., & Novità, A. (2015). Physically based modelling of soil erosion induced by rainfall in small mountain basins. *Geomorphology*, *243*, 106–115.

Desta, L., Carucci, V., & Wendem-Ageňehu, A. (2005). *Community-based Participatory Watershed Development: A Guideline PART 1. Ministry of Agriculture and Rural Development*. Retrieved from ⟨ftp://197.156.65.34/SNSF⟩ Administration and Finance Manuals/Administrative Materials/PSNP-HABP May 2011/May 2011 GoE Key Documents/Programme Implementation/Public Works/Nov-Dec, 2009 TOT/WSM Guidelines full text.January 2005.doc.

Gebrehiwot, T., & A, van der, V. (2014). *The Effect of Enclosures in Rehabilitating Degraded Vegetation : A Case of Forest Research*, *3*(*4*), 7. http://dx.doi.org/10.4172/2168-9776.1000128.

Gebremichael, D., Nyssen, J., Poesen, J., Deckers, J., Haile, M., Govers, G., & Moeyersons, J. (2005). *Effectiveness of stone bunds in controlling soil erosion on cropland in the Tigray Highlands, northern Ethiopia. Soil Use and Management*, *21*. http://dx.doi.org/10.1079/SUM2005321.

Gebresamuel, G., Singh, B. R., & Dick, Ø. (2010). Land-use changes and their impacts on soil degradation and surface runoff of two catchments of Northern Ethiopia. *Acta Agriculturae Scandinavica Section B–Soil and Plant Science*, *60*(*3*), 211–226.

Gessesse, G. D. (2009). *Modelling spatial rill initiation on rough soil surface using rill survey and digital photogrammetry at Angereb Watershed, Ethiopia*. Vienna: University of Natural Resources and Applied Sciences.

Grunder, M. (1992). *Results of soil conservation experiments in Ethiopia. Proceedings of second natural resources conservation conference*. Addis Abeba: Institute of Agricultural research.

Haregeweyn, N., Berhe, A., Tsunekawa, A., Tsubo, M., & Meshesha, D. T. (2012). Integrated watershed management as an effective approach to curb land degradation: A case study of the enabered watershed in northern Ethiopia. *Environmental Management*, *50*(*6*), 1219–1233.

Haregeweyn, N., Poesen, J., Nyssen, J., De Wit, J., Haile, M., Govers, G., & Deckers, S. (2006). Reservoirs in Tigray (Northern Ethiopia): Characteristics and sediment deposition problems. *Land Degradation and Development.*

Haregeweyn, N., Tsunekawa, A., Poesen, J., Tsubo, M., Meshesha, D. T., Fenta, A. A., Nyssen, J., et al. (2017). Comprehensive assessment of soil erosion risk for better land use planning in river basins: Case study of the Upper Blue Nile River. *Science of the Total Environment*, *574*, 95–108.

Herweg, K., & Ludi, E. (1999). The performance of selected soil and water conservation measures: Case studies from Ethiopia and Eritrea. *Catena*, *36*(*1*), 99–114.

Hurni, H. (1985). Erosion – Productivity – Conservation Systems in Ethiopia. *Proceedings IV International Conference on Soil Conservation* pp. 654–674. Maracay, Venezuela.

Hurni, H. (1998). *Agroecological belts of Ethiopia. Explanatory notes on three maps at a scale of 1: 1,000,000. Soil conservation research programme. research report 43* (p. 30) Addis Abeba and Bern: Centre for Development and Environment (CDE) (with map, scale1: 5 million).

Hurni, K., Zeleke, G., Kassie, M., Tegegne, B., Kassawmar, T., Teferi, E., Moges, A. et al. (2015). *Economics of Land Degradation (ELD) Ethiopia Case Study: Soil degradation and sustainable land management in the rainfed agricultural areas of Ethiopia: An assessment of the economic implications*. Water and Land Resource Centre (WLRC); Centre for Development and Environment (CDE); Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ).

Kaltenrieder, J., Hurni, P. D. H., & Herweg, D. K. (2007). Adaptation and validation of the Universal soil loss equation (USLE) for the Ethiopian-Eritrean highlands. *Centre for Development and Environment*.

Kassawmar, T., Eckert, S., Hurni, K., Zeleke, G., & Hurni, H. (2016). Reducing landscape heterogeneity for improved land use and land cover (LULC) classification over large and complex Ethiopian highlands. *Geocarto International*. (*0*)(0)(, 0)

Lee, G.-S., & Lee, K.-H. (2006). Scaling effect for estimating soil loss in the RUSLE model using remotely sensed geospatial data in Korea. *Hydrology and Earth System Sciences Discussions*, *3*(*1*), 135–157.

Lemenih, M., Olsson, M., & Karltun, E. (2004). Comparison of soil attributes under Cupressus lusitanica and Eucalyptus saligna established on abandoned farmlands with continuously cropped farmlands and natural forest in Ethiopia. *Forest Ecology and Management*, *195*(*1–2*), 57–67.

Mekuria, W., Veldkamp, E., Haile, M., Nyssen, J., Muys, B., & Gebrehiwot, K. (2007). Effectiveness of exclosures to restore degraded soils as a result of overgrazing in Tigray, Ethiopia. *Journal of Arid Environments*, *69*(*2*), 270–284.

Munro, R. N., Deckers, J., Haile, M., Grove, A. T., Poesen, J., & Nyssen, J. (2008). Soil landscapes, land cover change and erosion features of the Central Plateau region of Tigrai, Ethiopia: Photo-monitoring with an interval of 30 years. *Catena*, *75*(*1*), 55–64.

Nyssen, J., Poesen, J., Moeyersons, J., Deckers, J., Haile, M., & Lang, A. (2004). Human impact on the environment in the Ethiopian and Eritrean highlands – A state of the art. *Earth-Science Reviews*, *64*(*3–4*), 273–320.

Panagos, P., Borrelli, P., Meusburger, K., Alewell, C., Lugato, E., & Montanarella, L. (2015). Estimating the soil erosion cover-management factor at the European scale. *Land Use Policy*, *48*, 38–50.

Panagos, P., Borrelli, P., Meusburger, K., van der Zanden, E. H., Poesen, J., & Alewell, C. (2015). Modelling the effect of support practices (P-factor) on the reduction of soil erosion by water at European scale. *Environmental Science and Policy*, *51*, 23–34.

Renard, K., Foster, G., Weesies, G., McCool, D., & Yoder, D. (1997). Predicting soil erosion by water: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). *Agricultural Handbook No. 703.* Retrieved from ⟨http://www.ars.usda.gov/SP2UserFiles/Place/64080530/RUSLE/AH_703.pdf⟩.

Renschler, C. S., Mannaerts, C., & Diekkruger, B. (1999). Evaluating spatial and temporal variability in soil erosion risk–rainfall erosivity and soil loss ratios in Andalusia, Spain. *Catena*, *34*(*3–4*), 209–225 (Retrieved from)⟨http://www.sciencedirect.com/science/article/B6VCG-3VTYVWW-1/2/dafb83536029f0d36e8cd2e3c8996b17⟩.

SCRP (2000). *Long-term monitoring of the agricultural environment (1982–1994) in the area of andit tid, anjeni and Maybar, Soil Conservation Research Programm (SCRP), Ethiopia*. Switzerland: University of Bern.

Subhatu, A., Lemann, T., Hurni, K., Portner, B., Kassawmar, T., Zeleke, G., & Hurni, H. (2017). Deposition of eroded soil on terraced croplands in Minchet catchment, Ethiopian highlands. *International Soil and Water Conservation Research* (pp. 1–9), 1–9.