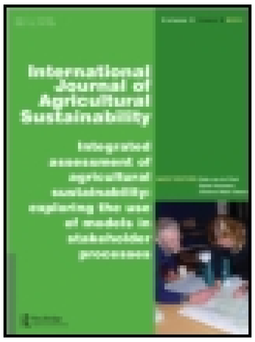

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Assessment of sustainable land use: linking land management practices to sustainable land use indicators

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ABSTRACT

Land degradation threatens food production especially in smallholder farming systems predominant in sub-Saharan Africa. Monitoring the effects of agricultural land uses is critical to guide sustainable intensification (SI). There are various indicators of sustainable land use (SLU), but conventional methods to quantify their metrics are complex and difficult to deploy for rapid and large-scale assessments. Considering that SLU indicators are dependent on agricultural practices, which can be rapidly identified and quantified, we propose a framework for SLU assessment that includes indirect quantifications of prioritized indicators (crop productivity, soil organic carbon (SOC), acidification, erosion, nutrient balance) using agricultural practices; and a SLU index derived from the integration of these indicators. The application of the framework to a case study, consisting of 1319 farm plots in Tanzania, reveals that SOC and N balance were the main contributors to the SLU gap. Only 2.2% of the plots qualified as being used sustainably. The framework proved to be sensitive to practices commonly used by farmers, thus providing an opportunity to identify practices needed to revert land degradation. Further application of the framework as a decision-support tool can enhance the efficiency of SI investments, by targeting practices which effectively enhance food production and preserve land.

KEYWORDS

Acidification; erosion; partial N balance; crop productivity; soil carbon; sustainable land use index

1. Introduction

Sustainable intensification (SI) is recognized as a key requirement to meet food demands for the rapidly growing population and preserve ecosystem services (Zurek et al., 2015). This is particularly important for sub-Saharan Africa (SSA) where food insecurity is prevalent (van Ittersum et al., 2016) and safety nets are weak for agricultural production. Despite broad support for SI, significant gaps persist in crop yields and overall production under smallholder farming systems in SSA (Tittonel & Giller, 2013; van Ittersum et al., 2016), and the impact of agricultural practices on natural resources is often considered as a secondary objective. Sustainable land use (SLU) implies that land is used to meet human needs while preserving key ecosystem services (Ghersa et al., 2002; Gutzler et al., 2015), therefore it is an integral component of SI (Pretty et al., 2011; Zurek et al., 2015). Land degradation, due to agricultural practices, can occur relatively rapidly, particularly in densely populated

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farming areas of SSA, while the process for reverting degraded lands can be long, if at all practical or possible (Scherr & Yadav, 1996; Willy et al., 2019). Processes of land degradation induced by agricultural practices include, among others, (i) the loss of nutrients mainly as a result of continuous cultivation without adequate replenishment of nutrients, but also through leaching and erosion (Kihara et al., 2015; Otsuka & Larson, 2013; Zhou et al., 2014); (ii) the loss of soil organic matter due to the conversion of land to agriculture, which subjects organic matter to rapid decomposition, insufficient addition of organic matter, and erosion (de Moura et al., 2016; Roose & Barthes, 2001); (iii) the loss of top soil through erosion due to the lack of soil cover and inappropriate tillage methods (Scherr & Yadav, 1996); (iv) the soil acidification due to the use of acidifying fertilizers, or losses of bases through continuous removal of crop residues (Avila et al., 2005; Randall et al., 2006); (v) the salinization due to accumulation of salts particularly in irrigated and semi-arid areas (Rengasamy, 2006); and (vi) the compaction mainly due to the use of machinery (Bately, 2009).

Various frameworks related to land use assessment have been developed, including the framework for evaluating sustainable land management (Smyth & Dumanski, 1993) and the sustainability assessment of agricultural systems (van Cauwenbergh et al., 2007). These frameworks have been summarized and compared by Acosta-Alba and van der Werf (2011). While they provide logical steps for making decision, their applicability is often limited by the lack of practical guidelines and non-specificity of indicators. Recently, frameworks were developed for sustainable intensification assessment, including a focus on smallholder farming systems (Musumba et al., 2017; Smith et al., 2017). The proposed approaches for assessing specific metrics related to SLU indicators in these frameworks are based on conventional methods, which use field and/or laboratory measurements. However, for large-scale and rapid monitoring, conventional methods are limited by high costs for intensive sampling and analysis, delays in delivering results due to time-consuming measurements, and the lack of accuracy of some of the available methods.

Indicators of SLU should be sensitive to a range of agricultural management practices. Management practices which are common within smallholder farming systems in SSA include, the use of inputs (manure and other organic inputs, fertilizers, biofertilizers such as inoculants), cultivation of nitrogen-fixing legumes, use of soil conservation practices (fallow, mulch, hedgerows), and tillage methods (Tittonell et al., 2010; Vanlauwe et al., 2010). These management practices can be relatively easily identified and quantified, and if the quantitative relationship between these practices and priority indicators can be established, then the relatively rapid monitoring of changes in those indicators is possible.

Therefore, the objectives of this paper were (i) to develop a framework for assessing SLU based on quantifiable relationships between agricultural management practices and the priority SLU indicator metrics and (ii) to evaluate the proposed framework based on a case-study in Tanzania.

In line with these objectives, the paper is step-wisely structured into five sections: First, we prioritize SLU indicators and their conventional quantification methods; Second, we propose indirect methods for linking their quantification to specific management practices; Third, we describe the process for aggregating the indicators in a single SLU index, including defining indicator thresholds and scores; Fourth, we apply the framework to a case-study in Tanzania; and finally we discuss the results and the applicability of framework.

2. Choice of SLU indicators

A wide range of indicators related to SLU has been proposed in various SI assessments (Musumba et al., 2017; Smith et al., 2017; COSA, https://thecosa.org). We opted to work around four principles, which we consider as critical in the context of smallholder farmers in Africa (FAO, 2015): (i) to keep the soil in place since losses of soil remove the most fertile topsoil; (ii) to increase or maintain soil organic C (SOC); (iii) to maintain adequate level of nutrients to avoid depleting soil nutrient stocks; and (iv) to avoid soil acidification. Other principles that are important to SLU are not included in this framework because, as far as we know, indirect methods to link these to management practices are virtually absent. These include (i) to avoid nutrient losses to the environment since leaching and gaseous losses can result in environment pollution and (ii) to increase or maintain functional soil biological diversity since soil biota govern several critical soil processes supporting the provision of soil-related services.

In line with the four principles cited above, four indicators were retained: Soil erosion, soil organic C, nutrient balance, and soil acidification. A fifth indicator (crop productivity) was included because productivity is indispensable to advance the relevance of SLU and achieve SI. Crop productivity is the most widely used indicator in sustainability assessment of agroecosystems (Smith et al., 2017), whereas the first four indicators are commonly included in assessments of soil quality and of the environmental impact of SI (FAO, 2014; Kanter et al., 2018; Lewandowski et al., 1999; Musumba et al., 2017; Raiesi & Kabiri, 2016; Sione et al., 2017; Smith et al., 2017).

2.1. Crop productivity

To overcome the prevailing food insecurity and the expected increase in food demand for the growing population, crop productivity must increase in SSA (van Ittersum et al., 2016). In our framework, crop productivity refers to the edible product harvested per unit area of land. Plant parts, other than the edible product, are considered for the assessment of other indicators, including SOC, nutrient balances and acidification. The direct measurement of yields usually consists of the destructive harvest of the crop at maturity for a known area of land. The use of this method for large-scale yield assessments in smallholder farmers' fields is constrained by associated high costs, given that a good representativeness of sampling plots is needed to cover the high within field variability prevailing in smallholder farms (Tittonel et al., 2007). That said, new methods based on the use of remote sensing, drone applications, and data science are being developed and validated though success in highly variable smallholder production areas has been limited (Burke & Lobell, 2017; Jain et al., 2016). In SSA, large-scale assessments of yields are generally obtained through surveys, where farmers are asked to report the quantity of crop produced from a given field or farm area. The main drawback of this recall method is the lack of accuracy, particularly associated with estimations of the surface area from which crops are harvested (Sapkota et al., 2016). Because of lack of appropriate alternatives, in this framework, we adopted the farmer recall approach, assuming that alternative indirect methods will become available in the near future.

2.2. Nutrient balances

Besides inherent soil properties, nutrient stocks are influenced by the nutrient inputs and outputs in the soil system (de Jager et al., 1998; Groppo et al., 2015; Wortmann & Kaizzi, 1998). Soil nutrient balances are defined as the difference between the amount of nutrients added to the soil (inflows) and the nutrients removed from the soil (outflows). When the nutrient inflows exceed the outflows (positive nutrient balance), the system accumulates nutrients, which can cause risks to natural resources, such as water pollution (Smith & Siciliano, 2015). When nutrient outflows exceed inflows (negative nutrient balance), nutrients in the system are depleting, which can be detrimental to crop productivity, particularly where there are insufficient soil nutrient stocks (Zhou et al., 2014). The direct assessment of nutrient balances requires measuring nutrient contents in all nutrient inflows to- and outflows from -the soil. However, the conventional measurements of the various nutrient flows (atmospheric deposition, gaseous losses, soil erosion and leaching) are difficult and modelling approaches are prone to various limitations, leading to inaccuracy, and high variations in estimates from different methods (Cobo et al., 2010; Sainju, 2017). Due to difficulties in measuring full nutrient balances, authors often report partial nutrient balance where inflows consist of the nutrients added to the soil through fertilizers, organic amendments and biological fixation of N, and outflows the nutrients removed from the soil through the harvest of crop produce and the removal of crop residues from the fields (Ngome et al., 2011; Vitousek et al., 2009; Zingore et al., 2007).

2.3. Soil organic C

The soil organic C (SOC) content is a critical indicator of soil health and influences various soil biological, chemical, and physical processes (Lal, 2015). Generally, soils in smallholder farming systems in SSA have low SOC contents, partly explaining their low productivity and vulnerability to degradation (Musinguzi et al., 2013; Willy et al., 2019). The addition of organic inputs is one of the main options to increase SOC content, though the magnitude of the effect does depend on the quantity and quality of organic inputs added, the physical and chemical environment, and the soil biotic activity (Fujisaki et al., 2018; Swift et al., 1979). The direct measurement of SOC is based on oxidation or combustion of the soil such as in the Walkley Black method or in CHN/CNS analysers. Due to the high spatial variability of SOC, intensive sampling and accurate analyses are required. Given that SOC is a large pool relative to the small changes occurring due to management, it is difficult to detect short-term changes in SOC content due to specific agricultural management practices (Adewopo et al., 2014; Dou et al., 2016).

2.4. Acidification

One of the major constraints to enhancing crop productivity in parts of SSA is soil acidity-related constraints to crop production (Vågen et al., 2016) while lime is seldom used, mainly because of high costs and limited promotion of the practice (Crawford et al., 2008). Factors contributing to soil acidification include, among others, the use of acidifying N-containing fertilizers and excess removal of cationic ions without their replenishment (Bolan et al., 1999; Goulding, 2016). While the direct measurement of pH is fast and cheap, assessing pH at large-scale and at adequate frequencies requires intensive and costly field campaigns.

2.5. Soil erosion

Soil erosion is an important issue in smallholder farming systems and its prevalence is mostly due to continuous cultivation, leading to non-replenished removal of soil cover and the expansion of agriculture to hilly areas, particularly in densely populated regions (FAO, 2015; Pimentel & Burgess, 2013). The scope of this framework is limited to soil erosion caused by water because this has extensively been studied and is prevalent in all agricultural systems, whereas wind erosion tends to be localized in semi-arid and arid areas (FAO, 2015). Removal of soil cover and inappropriate tillage methods are the major triggers of soil erosion, whereas soil conservation practices such as mulching and physical or biological barriers minimize it (Labriere et al., 2015; Wolka et al., 2018). Soil erosion due to water at field scale is usually assessed in Wischmeier plots by capturing water and soils at the bottom of the field for quantification (Wischmeier & Smith, 1978). The installation of Wischmeier plots is an expensive undertaking, whereas the frequent measurements require substantial logistics and time, making the method impracticable for rapid or large-scale assessments (Stroosnijder, 2005). Furthermore due to possible redistribution of soil among small areas within a field, which is not accounted for in Wischemeier plot, the method can underestimate or overestimate soil losses (Sanchez, 2019).

3. Relationships between SLU indicators and land management practices

Given the difficulties in measuring the proposed indicators rapidly or at scale with conventional methods, and the fact that the status of the priority indicators is influenced by specific management practices, we propose the use of indirect methods to assess metrics related to each indicator, based on documented relationships between the practices and those metrics. The following section describes the proposed indirect methods. This does not apply to the crop productivity indicator which is estimated directly from the quantity of harvested produce, as indicated in Section 2.1, above.

3.1. Partial N balances

We limited our assessment of nutrient balances to N, given that it is often the most limiting nutrient to crop production in SSA (Kihara et al., 2016; Liu et al., 2010). Some crops such as root crops and banana, require significant amount of potassium, therefore K becoming very important. However, those crops still require significant amounts of N to achieve high yields (Ezui et al., 2016).

N contents of organic inputs that are commonly used for soil management are available through the organic resource database (Palm et al., 2001) or other published documents (Sanginga & Woomer, 2009; Woomer et al., 1999; Zingore et al., 2014). The quantity of crop residues returned to-or removed from – field at harvest are derived from crop yields using harvest indices reported in literature (Musumba et al., 2017).

Biological N fixation (BNF) is a major input of N to cropping systems, contributing about half of the total crop N uptake in legumes (Giller et al., 1997; Ladha et al., 2015). However, its measurement is cumbersome and time-consuming (Unkovich et al., 2008). In this framework, we have indicated average values from the ranges of proportions of N derived from BNF reported in literature (Table 1) for each of the most common grain legumes grown under smallholder farming systems of SSA, which can be used when data are not available.

It is important to note that the proposed estimation of partial N balance does not take into consideration soil N stocks. For soil with large stocks of available nutrients, negative balances would be acceptable for a limited period, whereas positive balances can result in significant losses to the environment. However, most soils in smallholder farming systems are low in soil organic matter, therefore, large soil N stocks are uncommon (Tittonell et al., 2010; Zingore et al., 2007).

3.2. Soil organic C

The main sources of SOC related to management practices are organic inputs (e.g. manures, compost), roots, and crop residues (Dawson & Smith, 2007; Fujisaki et al., 2018; Raffa et al., 2015). For the indirect assessment, we adopted the quantity of *C* added to soil from these types of inputs, as the metric for SOC, and is referred to as C_{applied}. The quantity of C_{applied} is determined by multiplying the *C* content of the organic input with the quantity of organic input added to the soil. The *C* content in plant-derived inputs is estimated to be 45% (Mtangadura et al., 2017). The *C* content of manure is, however, very variable. Therefore, where available, it is recommended to use local estimates of *C* content in manure. However, in absence of local data, a *C* content of 28% in manure can be used (Larney et al., 2005; von Arb et al., 2020). For the contribution of the root biomass, the quantity of roots is estimated as 35% of the above ground biomass (AGB) as used in Woomer (2003), and later reported in Kamoni and Gicheru (2014) for various annual crops.

3.3. Acidification

Acidification generated from the use of fertilizers was assessed based on the amount of fertilizer applied and the acidifying potential of fertilizers (Table 2) as per Equation (1) (Bolan et al., 1999). Similar values based on the amount of N applied in fertilizers can be found in Hollier (1999) and Wortmann et al. (2015).

$$\text{Fertilizer Acid Generation} = \text{Acidifying potential} \times \frac{\text{total amount of fertilizer (kg)}}{100} \quad (1)$$

Table 1. Fraction (%) of total N that is derived from Biological Nitrogen Fixation (BNF) for legume species commonly grown in sub-Saharan Africa.

Crop species	Fraction of N from BNF (%)	Average
Common bean	10–51[a]	30
Soybean	35–89[a,b]	62
Groundnut	38–62[a]	50
Cowpea	8–89[a]	48
Pigeon pea	4–88[a]	46

[a]Giller et al. (1997); [b]van Vugt et al. (2018).

Acidification due to the removal of excess base (EB) is estimated following Equation (2).

$$\text{Excess Base Acid Generation} = 0.05 \times \frac{\text{EB}}{100} \times \text{total amount of residue (kg)} \quad (2)$$

where 0.05 is the amount of $CaCO_3$ required to neutralize the acidity generated by one mole of EB (Havlin et al., 1999); EB is the amount of excess base in one kg of plant residue (cmol kg^{-1}); and 100 refers to the conversion of EB from cmol kg^{-1} to mol kg^{-1}. Estimates for excess bases for some crop residues that are relevant to SSA are available (Table 3) and can be used in Equation (2) when specific data are not available.

The buffer capacity (BC) of a soil is a major property influencing pH change. Soils with high BC require more protons to reduce the pH by one unit. In general, coarse soils will have a lower BC than heavier soils (Hollier, 1999). In absence of BC measurements, the coefficients developed in Hollier (1999) can be used: 2, 3, 4, 9, respectively for sand, sandy loam, sandy clay loam; and clay soils. The acid generated by the practices is subtracted from the pH at the location (baseline pH) to obtain the actual pH, which is used in scoring the acidification indicator as defined under the 'Indicator thresholds and scores' section below. Where pH data are not available, the baseline pH can be obtained from ISRIC soil grids (https://soilgrids.org), though the current information is at 250 m spatial resolution. Ongoing efforts to produce the information at 30 m resolution could improve the accuracy, once it is finalized.

3.4. Soil erosion

In this framework, we adopted the Revised Universal Soil Loss Equation (RUSLE) (Equation 3), which consider management practices that farmers use such as soil conservation practices, tillage methods, soil cover, in

Table 2. Selected fertilizer products used in sub-Saharan Africa and their acidifying potential.

Fertilizer type	Acidifying potential[a]
Ammonium sulfate	110
Diammonium phosphate (DAP)	74
Elemental sulfur (S)	310
Single superphosphate (SSP)	8
Urea	79

Source: Bolan et al. (1999).
[a]kg $CaCO_3$ required to neutralize the acid generated with 100 kg of fertilizer.

Table 3. Amount of excess base in residues of selected crops grown in sub-Saharan Africa.

	Excess base (cmol kg^{-1})		
Crop	Average	Min	Max
Wheat	30	25	37
Sorghum	37	29	44
Barley	38	26	49
Maize	56	38	75
Soybeans	117	106	143
Cabbage	55	38	79
Lettuce	120	114	125
Carrot	128	108	139
Tomato	164	128	184
Spinach	197	190	208

Adapted from Pierre and Banwart (1973).

addition to biophysical features. Various studies have applied the equations in SSA, using different estimates for the parameters in the equations (Karamage et al., 2016a; Tamene & Le, 2015). Although the model does not provide precise quantification of soil losses, it is widely used for comparing systems and monitoring changes in erosion trends (Gutzler et al., 2015):

$$A = R*K*LS*C*P \tag{3}$$

where A is the soil loss per unit area (Mg ha^{-1} yr^{-1}), R is the rainfall erosivity factor (MJ mm hr^{-1} ha^{-1} yr^{-1}), K is the soil erodibility factor (Mg hr MJ^{-1} mm^{-1}), L stands for the slope length factor, S for the slope-steepness factor, C represents the cover management factor and P is the support practice factor.

Various equations have been proposed to estimate R, K, and LS for different contexts. Equations used in this framework are indicated in Table 4. The C factor has been shown to correlate with NDVI values (De Jong et al., 1999). Therefore, a spatially – explicit C factor can be derived by assessing relationships between remotely-sensed NDVI values and corresponding C factor values obtained from USLE/RUSLE guide tables or computed from field observation (Karaburun, 2010). However, given that NDVI indicates the greenness of a vegetation and not its structural characteristics, and that the use of free NDVI products is limited by the coarse spatial resolution (250 m^2/pixel), NDVI may not be useful for smallholder farming systems, where the unit farmland is very small (<1 ha) and where crop types change within a short distance. In the absence of NDVI, literature values of C factor such as reported in Morgan (2005) can be used (Table 5).

Table 5. Cover management factor C for various land cover and for specific crops.

Land cover/crops	C factor	Mean C factor
Bare ground	1	1
Pasture	0.1	0.1
Bananas	0.1–0.3	0.2
Cocoa	0.1–0.3	0.2
Coffee	0.1–0.3	0.2
Wheat	0.1–0.4	0.3
Cotton	0.4–0.7	0.6
Maize	0.2–0.9	0.4
Groundnut	0.3–0.8	0.6
Palm with cover crops	0.1–0.3	0.2
Potatoes	0.1–0.4	0.3
Soybean	0.2–0.5	0.4
Yams	0.4–0.5	0.5

Adapted from Morgan (2005).

The P factor is related to soil conservation practices (Adornado et al., 2009). It reflects the effects of practices on reducing the amount and rate of runoff water. The most used support practices in cropland are the types of tillage, the use of contouring, terracing, mulching and hedges. A proper determination of the P factor would require plots of 1 ha (Roose, 1977). Where data are not available, P factor values from literature (Table 6) can be used.

Table 4. Approaches used to estimate the R, K, and LS parameters in the RUSLE equation to estimate soil losses.

Parameter	Approach used and reference	Estimates of parameters in approach	Potential Source of data
R	MFI-based for Africa: Vrieling et al. (2010) $R = 50.7*\mathrm{MFI} - 1405$	MFI Arnoldus (1977) $\mathrm{MFI} = \frac{\sum_{i=1}^{12} pi^2}{P}$ pi = precipitation of month i, P = annual precipitation	CHIRPS merged datasets between ground station and satellite data
K	Texture based Römkens et al. (1997) $K = 0.0034 + 0.0378*e^{-0.5((\log Dg + 1.533)/0.7671)^2}$ Dg is the geometric mean diameter of the soil particles (mm)	$Dg = e^{\left(0.01*\sum_{i=1}^{n} fi*\ln mi\right)}$ fi denotes the weight percentage of the particle size fraction (%); mi denotes the arithmetic mean of the particle size limits (mm); and n is the number of particle size fractions considered	ISRIC or AFSIS soil grids https://soilgrids.org.
LS	Breetzke (2004); Karamage et al. (2016a, 2016b) $LS = \left(\frac{X}{22.3}\right)^n (0.065 + 0.045S + 0.0065S^2)$	X = slope length (m); S = slope gradient (%); n = 0.5 for slope >5%, 0.4 for slope 3.5–4.5%, 0.3 for slope 1–3.5%, 0.2% for slope <1%.	Digital Elevation Model (DEM) from STRM or ASTER

4. Aggregation of indicators

To assemble the various effects of management practices on land use, we proposed a sustainable land use index (SLU index), based on the five indicators described above. Since the indicators are in different units, it is imperative to scale these so that they can be submitted to calculus, as commonly done in sustainability assessments (Carraro et al., 2009; Vasu et al., 2016). We adopted a scaling approach whereby indicator values were converted to scores (Carraro et al., 2009; Raiesi & Kabiri, 2016), in reference to specific indicator thresholds values as described below. For this assessment, we specify that a 10 years' timeframe is required to implement effective management practices and reverse the current land degradation trend. Otherwise, if no remediation occurs during that period, the degradation will become severe and have detrimental effects. The Africa Group of Negotiators Experts Support (AGNES) (2020) projects that if land degradation continues at the current pace, Africa may be able to feed just 25% of its population by 2025, and that more than half of the cultivated agricultural area could become unusable by 2050. The timeframe is also in line with the Sustainable Development Goals, which are expected to be achieved by 2030, therefore requiring good progress to be made in the next 10 years.

4.1. Indicator thresholds and scores

For each indicator, we set a threshold value (or range) considered as satisfying for the sustainability objective. We then converted values of each indicator into scores varying between 0 and 1 in reference to the threshold value. A score of 1 means that the indicator has reached the minimal desirable value of the indicator that would be required for SLU. A score of 0 represents an indicator value that results in unsustainable land use. Scores between 0 and 1 represent transitionary situations where sustainability goals are at risk in the longer-term.

Table 6. Support practice factor *P* for various types of tillage and erosion control practices.

Tillage and Residue management/soil conservation practices	*P*-factor	Mean value of *P*-factor
Conventional tillage	1.00[a,b]	1.00
Zoned tillage	0.25[a]	0.25
Mulch tillage	0.26[a]	0.26
Minimum tillage	0.52[a]	0.56
Bench terraces	0.12–0.2[b]	0.15
Contour cropping	0.15–0.50[b]	0.30
Hedges	0.1–0.3[b]	0.2
Cover crop	0.1–0.5[b]	0.4

[a]David (1988), [b]Roose (1977).

4.1.1. Crop productivity

We define the crop productivity thresholds with reference to the attainable yields in rainfed conditions, referred to as water-limited yield potential (www.yieldgap.org). Using maize as an example, the attainable yields reported in SSA range between 6 and 11 Mg ha^{-1} (www.yieldgap.org). To achieve food self-sufficiency in SSA by 2050, ten Berge et al. (2019) estimated a need to increase maize yields from the current 20% of water-limited yield potential (i.e. attainable yields) to 50–75%. Fifty percent of attainable yield is 3 ton ha^{-1} for the low attainable limit. Various studies have shown that a quick increase in maize yields to 3 ton ha^{-1} or slightly more can be achieved if farmers apply a combination of fertilizers, improved varieties and good agronomic practices (Denning et al., 2009; Kihara et al., 2015; Nziguheba et al., 2010). This level of yield has also been defined as enough for a household to cover basic food needs and therefore set as the first mark towards the Africa Green Revolution (Sanchez, 2010). We therefore adopted 50% of the attainable yield as the threshold for the crop productivity indicator. Attainable yields of crops for which the information is not available on the yield gap website can be obtained from other literature sources, including Tittonel and Giller (2013) (various crops), Hillocks R (2014) (cassava), Norgrove and Hauser (2014) (banana). Given that attainable yields could vary with context, it is preferable to use locally relevant information. Thus, for any given crop, yields of 50% attainable yield or more receive a score of 1; for yields below the threshold, scores are linearly decreasing up to 0 for zero yields (Figure 1A).

4.1.2. Partial N balance

In nutrient depleted soils as found under most smallholder farming systems in SSA, it is crucial to replace at least the N removed by crops, whereas a supply of more N than that removed by the crop is needed to contribute to building the soil N reserve, though this supply should be in acceptable limits to avoid unintended harm to ecosystems. We defined the N balance threshold based on the traffic light scheme for N surplus in cropping systems developed by the EU Nitrogen expert Panel (2015) and applied in

Marinus et al. (2018) for sustainability assessment of smallholder farming in Ghana and Kenya. In that scheme, a N surplus target of 50–80 kg N ha^{-1} yr^{-1} in cropping systems is defined as modest and has a green light. We adopted that range as the threshold. Nitrogen balance values in that range received a score of 1. The scores for N balance values between 0 and 50 kg N ha^{-1} yr^{-1} increase linearly while those above 80 kg N ha^{-1} yr^{-1} decrease linearly (Figure 1B). N balances below 0 (mining) or beyond 120 kg N ha^{-1} (high N accumulation) obtain a score of 0.

4.1.3. Soil organic carbon

Notwithstanding several attempts to define a SOC threshold for tropical soils, there is currently no consensus, most likely because thresholds are dependent on texture and clay mineralogy while the functions that are regulated by SOC vary between soil types (Musinguzi et al., 2013). Therefore, we did not attempt to base our scoring on a threshold of SOC content. Rather, we based the threshold on the quantity of *C* applied to the soil. Relative to the 3 ton ha^{-1} maize yield that was established as threshold for the crop productivity indicator, we estimated that an equal quantity (3 ton ha^{-1}) of maize stover is produced based on a harvest index of 0.5 (Gaiser et al., 2010). The associated quantity of roots is estimated to be 2.1 ton ha^{-1} (Woomer, 2003). Using the *C* content of 45% in crop residues, the quantity of $C_{applied}$ to the soil by the stover and roots from a 3 ton maize grain, is estimated to be 2.3 ton C ha^{-1}, which is used as the threshold for SOC.

To determine the minimum $C_{applied}$ that would be required to avoid losses of SOC, we established a relationship between annual changes in SOC and annual quantities of $C_{applied}$ to soil from organic inputs, using studies conducted in SSA, where changes of SOC could be generated at specific time scales after the addition of specified quantities of organic inputs (Figure 2). Only studies conducted for at least 4 years were considered. From the relationship, we estimated a $C_{applied}$ of 1.4 ton C ha^{-1} yr^{-1} as the minimum required to initiate any build up. Below this quantity, losses of SOC are likely to occur. Although we found a limited number of studies conducted in SSA that reported all needed information for this estimation, the threshold is close to the 1.5 ton C ha^{-1} yr^{-1} reported in Fujisaki et al. (2018), where studies conducted in tropical croplands from various regions, mainly Latin America,South Asia and a few from Africa, were considered. For the scoring, $C_{applied}$ of 2.3 ton C ha^{-1} yr^{-1} and above receive a score of 1, whereas $C_{applied}$ of 1.4 ton C ha^{-1} yr^{-1} or below receive a score of 0 (Figure 1C). Between 1.4 and 2.3 ton C ha^{-1} yr^{-1} scores increase linearly.

4.1.4. Soil acidity

Several studies recommended a threshold pH value of 5.5 in water (Havlin et al., 1999) to minimize Al toxicity for soils with relatively large reserves of Al. Below that pH, a decrease in availability of some nutrients (e.g. P) and Al toxicity starts developing (Crawford et al., 2008). A pH of 7.5 is considered as the upper limit, beyond which alkalinity associated problems, such as limited availability of P, Boron and Zinc, develop (Crawford et al., 2008). We therefore adopted a pH range of 5.5 to 7.5 as threshold, qualifying for a score of 1. When pH goes below 4 or above 9, crops are severely affected, and except for highly acidity or alkalinity tolerant crops, most crops will not grow. For those pH values, a score of 0 is allocated. Between pH values of 4 and 5.5, scores increase linearly, whereas between values of 7.5 and 9, scores decrease linearly (Figure 1D).

4.1.5. Soil erosion

An absence of erosion should be the goal of SLU. However, in a short to medium term, some erosion could be allowed as management practices are put in place to eliminate it. Ideally, allowable erosion thresholds would be different for different soil types, based on their vulnerability to erosion with less erosion allowed in shallow soils compared to deeper soils (Bhattacharyya et al., 2008). Soils on steep slope would also be more vulnerable, therefore allowing only little room for soil loss. We adopted a threshold of 5 ton ha^{-1} yr^{-1}, based on the erosion classification by Haregeweyn et al. (2017), where RUSLE erosion rates of 0–5 ton ha^{-1} yr^{-1} were classified as very slight in Ethiopia. The threshold can be adjusted based on local information on soil type and slope. Soil losses of below or equal to 5 ton ha^{-1} yr^{-1} get a score of 1. Soil losses of 50 ton ha^{-1} yr^{-1} have been categorized as very severe, even for less vulnerable soils (FAO, 2015; Haregeweyn et al., 2017); we therefore used it as the benchmark for unsustainable land use, qualifying for a score of 0. For soil losses between 5 ton ha^{-1} yr^{-1} and 50 ton ha^{-1} yr^{-1}, scores decrease linearly (Figure 1E).

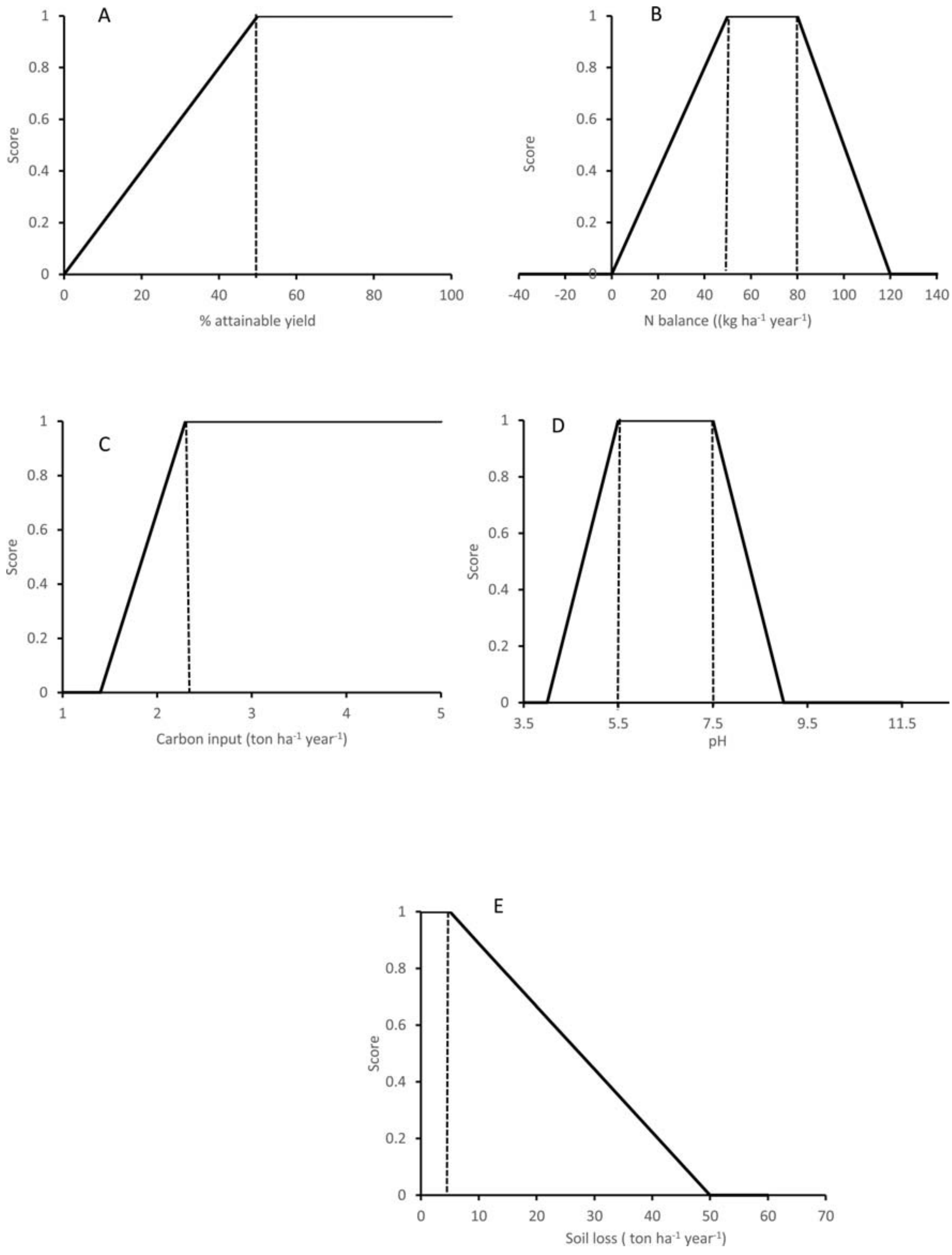

Figure 1. Trends of scores of the indicator values with reference to the defined threshold. A: Crop productivity, B: N balance, C: soil organic C, D: soil pH (acidification indicator), E: erosion. Vertical dash lines indicate the threshold values.

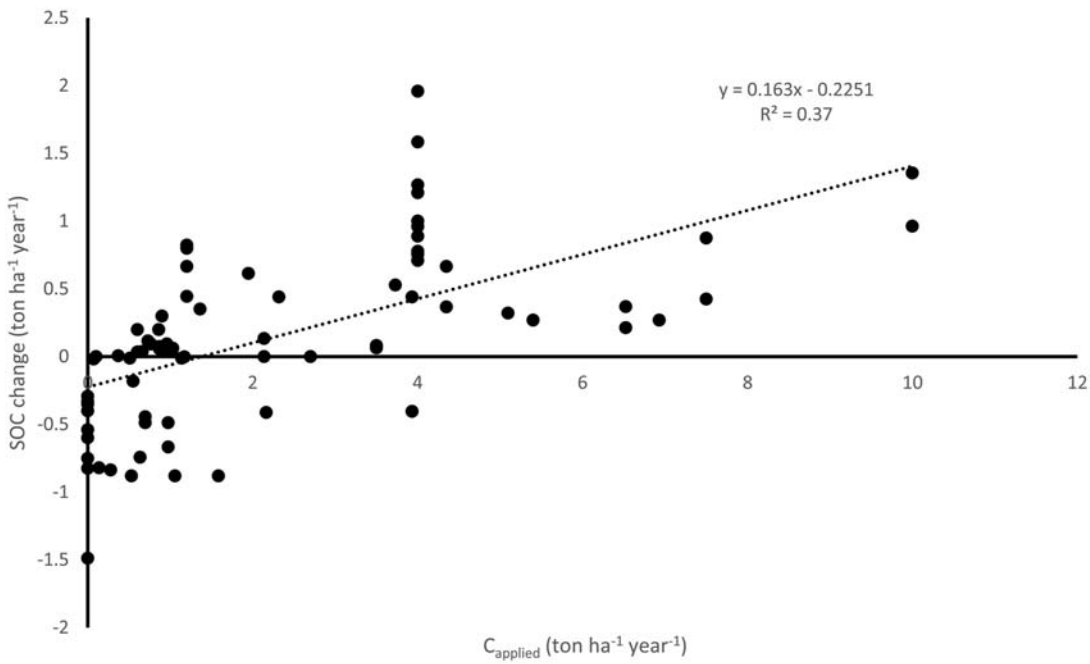

Figure 2 . Annual SOC change as a function of annual *C* applied to soil in organic inputs from various studies conducted in sub-Saharan Africa. Source: Akinnifesi et al., 2007; Bedada et al., 2014; Barthes et al., 2004; Diels et al., 2004; Fonte et al., 2009; Kamoni et al., 2007; Kapkiyai et al., 1999; Kihanda, 1996; Makumba et al., 2007; Mtangadura et al., 2017; Nakamura et al., 2011; Siguhara et al., 2012; Tojo Soler et al., 2011; von Arb et al., 2020.

4.2. Defining a composite sustainable land use index

The aggregation of indicators to generate a status index has often been done by summing the individual indicator scores, either directly or after weighting them (Laishram et al., 2012; OECD, 2008; Roboredo et al., 2016). While weighting is useful to compensate perceived differences in the importance between indicators towards the objective being assessed, there is often no standard method to allocate the weights. Subjective methods have therefore been used, based on individual or group judgement (Korbel & Hose, 2017; Shriar, 2000; Vasu et al., 2016). Such subjective methods can be biased, thus potentially compromising the results/inferences (Areal et al., 2018). To avoid the uncertainties around weighting and the likelihood of bias and considering that all the indicators that are prioritized in this framework are important for SLU, we opted to allocate an equal weight to all the indicators.

The direct summation of scores (weighted or not) is a well adopted method of aggregation (McCune et al., 2011; Sione et al., 2017). However, we opted not to use it because the resulting score can mask serious problems in an individual indicator, that would render a land use not sustainable. For example, with the summation of indicators, two land uses, one with indicator scores of 0.7, 0.3, 0.6, 0.4, 0.5 and the other of 0.6, 0, 0.5, 1.0, 0.4, would both give a SLU index of 0.5. However, a score of 0, as in the second case, is not acceptable for any of the indicators given that such a score renders land use unsustainable. We therefore proposed that the SLU index is equal to the lowest score among the indicators. This is because the indicator with the lowest score is expected to limit most the performance of the whole land (law of minimum). By this approach, the SLU index of a given land is equal to the lowest score among the indicators, as applied earlier in agriculture sustainability assessments (Zahm et al., 2008). In this framework, a land is sustainably used when all indicators have reached at least their threshold levels, implying that the SLU index equals 1.

5. Application of the framework to a case study in Tanzania

To test the applicability of the framework, we chose a project, which covers a range of biophysical conditions, in which diverse management practices are included, and in which plot level information was collected. The case study was not meant to be a formal impact study but rather to ascertain the sensitivity of the framework to management practices. The framework was applied to a project entitled 'Enhancing partnership among Africa RISING, NAFAKA and TUBORESHE CHAKULA programs for fast-tracking delivery and scaling of agricultural technologies in Tanzania, also referred to as Africa RISING-NAFAKA, a USAID-funded project under the Feed the Future program (https://africa-rising.net/category/partners/nafaka, IITA, 2018). The project focuses on delivery and scaling of promising interventions that enhance agricultural productivity in cereal-legume farming systems in Tanzania. Key interventions related to land use included the dissemination of best-bet crop management technologies (improved seeds, nutrient sources, rotation of legumes and cereals), and the rehabilitation and protection of natural resources (use of soil conservation approaches, soil erosion control). The project started in 2014, was being implemented in seven regions of Tanzania (Manyara, Dodoma, Njombe, Morogoro, Iringa, Mbeya and Songwe), and included a total of 9 districts and 35 wards. Training sessions were conducted for farmers and extension staff on the promoted technologies, using demonstration trials. In 2018, a survey was conducted to assess the level of use of the technologies in the areas covered by the project, including a module allowing for assessing the SLU.

5.1. Sampling and data collection

Farmers included in the survey were selected from a population of 35,855 beneficiaries. A systematic probability-proportional-to-size random sampling strategy as suggested for USAID-funded Feed the Future program interventions was used (Stukel & Friedman, 2016, Chapter 9). As per this strategy, villages were randomly selected, and a given number of households assigned to each village depending on the number of beneficiaries reached in each village, thus constituting the proportional-to-size sample as shown in Table 7. Households were purposively selected with the criterion being the involvement of the household in the production of maize, rice or legumes as the focus crops for the Africa RISING-NAFAKA project. In the end, a total of 608 farmers were selected for the survey. The survey was conducted from 11th to 28th September 2018, using a structured questionnaire for the interviews (Supplement 1). A team of 10 enumerators (3 female, 7 male) collected the data, using the IT-based KoboCollect tool, each covering about 4 respondents per day. Each interview session, including both the SLU questions and other questions needed for the project, lasted 70 minutes on average. For the SLU assessment, information on land management during the November 2017 to March 2018 cropping season was collected for each plot in the farm of the surveyed household (see Section 5 of supplemented survey). Collected information could be categorized as: plot characteristics (size, soil colour, slope), crop related (main crop grown, varieties, cropping system, proportion of the plot on which the main crop was grown), input type and quantity (fertilizer, organic, lime, inoculant), soil conservation measures to control erosion, harvest (quantity of harvest), and the management of crop residues after harvest (whether residues are left on the plot or are removed). The management practices of a total of 1319 plots were captured.

All data analyses were done with STATA 14.2, StataCorp 1985–2015.

5.2. Assessment of SLU indicators

5.2.1. Crop productivity

The main crops grown in the covered area were maize (50% of the surveyed plots), rice (22% of the plots) and

Table 7. Distribution of the households surveyed in the case study in Tanzania.

Region	District	Number of villages benefitting from project	Number of villages randomly selected	Total number of households benefitting from project	Number of Households surveyed
Morogoro	Kilombero	10	4	3889	65
Iringa	Iringa rural	30	3	3860	61
	Kilolo	28	6	6403	102
	Mufindi	10	2	2084	36
Njombe	Wanging'ombe	10	2	1556	29
Mbeya	Mbarali	20	8	8718	126
Songwe	Mbozi	25	6	6300	100
	Momba	21	5	5290	89
Total		154	36	38,100	608

common beans (13% of the plots). Other crops, including groundnut were grown as main crop on less than 7% of the plots. For each crop grown, the quantity of produce harvested in a plot where it was grown was reported by farmers in local units, then converted to kg by the enumerators during the survey. This quantity and the area on which the crop was grown in the plot were used to generate the crop yield in ton ha^{-1} as the metric for crop productivity. The quantities of produce were reported as dry weight, except for tuber crops for which they were reported as fresh weight.

5.2.2. Partial N balance

N added with fertilizer was calculated based on the quantity of fertilizer and the N concentration in the fertilizer. Di-ammonium phosphate (DAP) (basal: 18% N) and urea (Basal or top dress: 46%) were the main fertilizers used (35 and 25% of plots respectively). Other fertilizers NPK (15% N) Calcium Ammonium Nitrate (CAN) (21% N), and Yara (23% N) fertilizers were used in a few plots. More than half of plots (59%) did not receive any fertilizers. N added with organic inputs was calculated from the quantity of the organic inputs applied to the plot and their estimated N concentration (Zingore et al., 2014). N added from BNF was estimated based on the estimation of proportion of N derived from BNF in Table 1. N removed was calculated as the N uptake in the human edible part, which constitutes the harvested part of the crop. Where crop residues were removed from the field after harvest, the N uptake in crop residues was also added to the quantity of N removed.

5.2.3. SOC

Crop residues and animal manure were the two types of organic inputs that farmers reported to use mainly on their plots, each being applied on at least 10% of the plots identified in the survey. During the interview, farmers reported the quantity of organic inputs applied to each plot, which the enumerators converted to kg. The root biomass of crop grown on each plot was estimated as indicated in Section 3.2. The C_{applied} from the application of these organic inputs were calculated using a C content of 45% for crop residues, and 28% for manure.

5.2.4. Acidification

The acidification from fertilizers were calculated following Equation (1), using fertilizer acidifying potential in Table 2. Where crop residues were removed from the plot after harvest, acidification due to the removal of residues was estimated with Equation (2) based on crop specific excess base in Table 3. Given that we did not have pH data for each plot identified in the survey, we obtained pH data from the ISRIC soil grids database (https://soilgrids.org) and used the average pH for each of the 35 wards included in the study as baseline pH of all the plots within the ward. We then calculated the actual pH, resulting from the management practices, by subtracting the acidification due to fertilizer use and crop residue removal from the baseline pH.

5.2.5. Erosion

Soil losses were estimated as per Equation (3), with parameters R, K estimated as described in Table 4. Monthly and annual precipitations for the estimation of R were obtained at ward level from CHIRPS merged datasets between ground station and satellite data (ftp://chg-ftpout.geog.ucsb.edu/pub/org/chg/products/CHIRPS-2.0). Particle size in K estimations were obtained from ISRIC soil grids database (https://soilgrids.org). During the survey, information on slope level of each plot was collected in 4 categories: flat, gentle, moderate, steep slopes, which we allocated slope gradients of 1%, 5%, 10%, 15% in this assessment following FAO (2006) slope classification. Values in Tables 5 and 6 were used respectively for C and P factors, based on the crops, and soil conservation practices applied to the plot.

5.3. Results

For each of the indicators, results are reported for all the 1319 plots and for plots that received a combination of fertilizer, improved seeds, organic inputs referred to as ISFM (77 plots), and plots that did not receive any of the 3 practices (290 plots). The remaining 952 plots received at least one of these practices. For some indicators, results are also indicated for plots on which specific practices, known to influence most the indicator being considered, were applied (e.g. fertilizers for N balance; organic inputs for C_{applied}, Soil conservation practices for erosion).

5.3.1. Crop productivity

Yields varied between crops, with a mean of 0.7 ton ha^{-1} for common bean, 3.2 ton ha^{-1} for rice. Given the difference in productivity ranges for different crops, maize data are presented in

Figure 3A and are discussed in detail for yields. The productivity scores, however, include all crops given that the range is the same for all crops (0–1). For maize, the overall productivity ranged from 0 to 7.7 ton ha^{-1} with a mean of 2.1 ton ha^{-1} (Figure 3A). Yield ranges in plots with ISFM were larger (0–7.6 ton ha^{-1}) than that in plots without ISFM (0–3.1 ton ha^{-1}), with means of 2.9 ton ha^{-1} and 1.0 ton ha^{-1}, respectively. The threshold of 3.0 ton ha^{-1} for maize was reached in 24% of the plots. The productivity score, considering all crops and plots ranged from 0 to 1, with an overall mean of 0.59. A proportion of 26% of the plots attained a score of 1 (Figure 4). The mean productivity score was 0.47 for plots without ISFM, and 0.76 for plots on which ISFM was applied.

5.3.2. Partial N balance

The mean N balance from all plots was 8.5 kg N ha^{-1} ranging from -176 to 180 kg ha^{-1} (Figure 3B). In plots where fertilizers were applied, the mean N balance was 32 kg N ha^{-1}, whereas a negative mean of -21 kg N ha^{-1} was observed in plots that did not receive fertilizers. A large proportion of the plots (61%) had a N balance of 0 or negative. Plots on which ISFM was applied had N balance ranging from -45 to 169 kg N ha^{-1} with a mean of 38 kg N ha^{-1}, and all those without ISFM had a negative or zero N balance (Figure 3B). Overall, 7% of plots had a N balance score of 1, whereas 65% had a score of 0 (Figure 4). Using ISFM resulted in higher N balance scores compared to plots without ISFM (Figure 5A).

5.3.3. SOC

The overall quantity of $C_{applied}$ ranged from 0 to 9.8 ton ha^{-1} yr^{-1} with a mean of 1.3 ton ha^{-1} yr^{-1} (Figure 3C). The mean $C_{applied}$ in plots where organic inputs were applied was 1.5 ton ha^{-1} yr^{-1}. The mean $C_{applied}$ in ISFM plots was twice (1.8 ton ha^{-1} yr^{-1}) the quantity in plots without ISFM (0.9 ton ha^{-1} yr^{-1}, Figure 3C). The threshold value of 2.3 ton ha^{-1} yr^{-1}, was achieved in 17% of the plots. The overall mean score for $C_{applied}$ was 0.24. The mean score in plots which received organic inputs (0.27) was slightly higher than plots that did not receive organic inputs 0.23. Similarly to the crop productivity scores, the use of the ISFM improved the $C_{applied}$ score (0.38) compared to plots where ISFM was not applied (0.15) (Figure 5). Low scores (0) for $C_{applied}$ were observed in 67% of the plots (Figure 4), whereas 17% of the plots achieved a score of 1.

5.3.4. Acidification

The baseline pH across the study area was ≥ 5, with 83% of plots having a baseline pH of at least 5.5. The actual pH, after consideration of the practices applied to plots, ranged from 4.6 to 6.3, with an overall mean of 5.7 (Figure 3D). Therefore, none of the plots had a pH score of 0. The pH score ranged from 0.38 to 1, with a mean of 0.97. The mean of pH values and pH scores were similar between ISFM and non-ISFM plots (Figures 3D and 5).

5.3.5. Erosion

There was a wide difference in soil losses between plots (0.01-127 ton ha^{-1} yr^{-1}) with very few plots (1%) showing a soil loss above 75 ton ha^{-1}. The overall mean of soil loss was 10.2 ton ha^{-1} yr^{-1}. Soil losses were below the threshold (5 ton ha^{-1} yr^{-1}) in 60% of the plots (Figure 3E), with 48% being on the flat slope category. The mean soil loss in plots where no soil conservation practices (SCP) were applied was 12 ton ha^{-1} yr^{-1} whereas it was 6.6 ton ha^{-1} yr^{-1} where soil conservation practices were applied. The majority (74%) of the plots on slope categories other than flat had a soil loss >5 ton ha^{-1} yr^{-1}. Considering only plots on moderate slope (21% of plots), the mean soil loss was 15 ton ha^{-1} yr^{-1} when SCP were applied and 36 ton ha^{-1} yr^{-1} when no SCP were applied, with respective mean scores of 0.76 and 0.48. Soil erosion score ranged from 0 to 1, with a mean score of 0.87 (Figure 4). Using SCP as a component of ISFM on moderate slopes resulted in a higher erosion score (0.91) compared to plots where no SCP nor ISFM practices were applied (0.47) (Figure 5B).

5.3.6. SLU index

The overall SLU index ranged from 0 to 1 with a mean of 0.09 (Figure 4). Most of the plots (85%) had a SLU index of zero, mainly due to the N balance and SOC scores, which was 0 in 65% and 67% of the plots, respectively. A SLU index of 1, was achieved in 2.2% of the plots distributed across 6 of the 8 districts, therefore predesignated as 'Sustainable' based on our definition of sustainable land use (Figure 4).

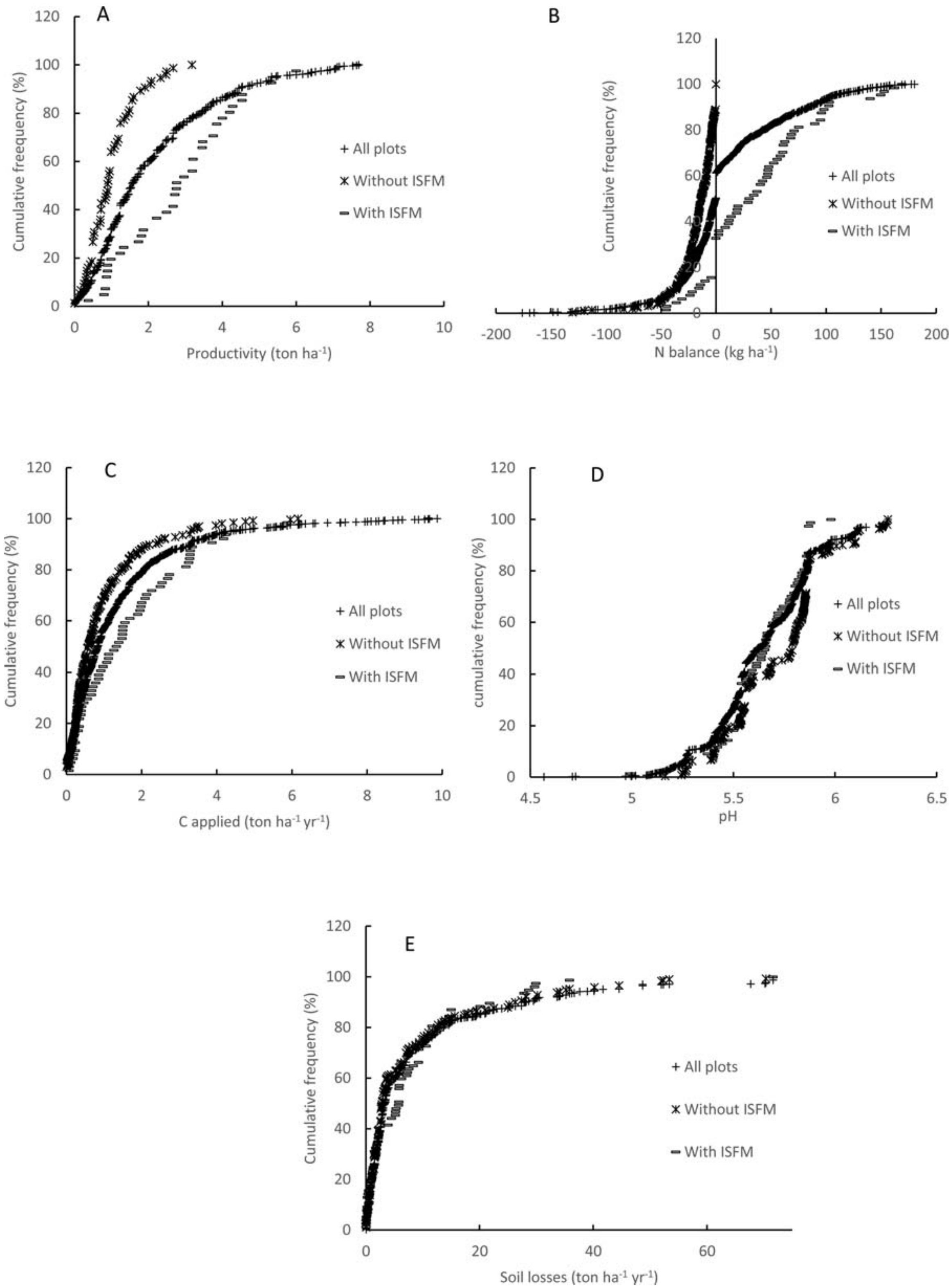

Figure 3. Frequency distribution of indicator values in the Tanzania case study. A: Crop productivity for maize, B: N balance, C: soil organic C, D: soil pH (acidification indicator), E: soil losses through erosion.

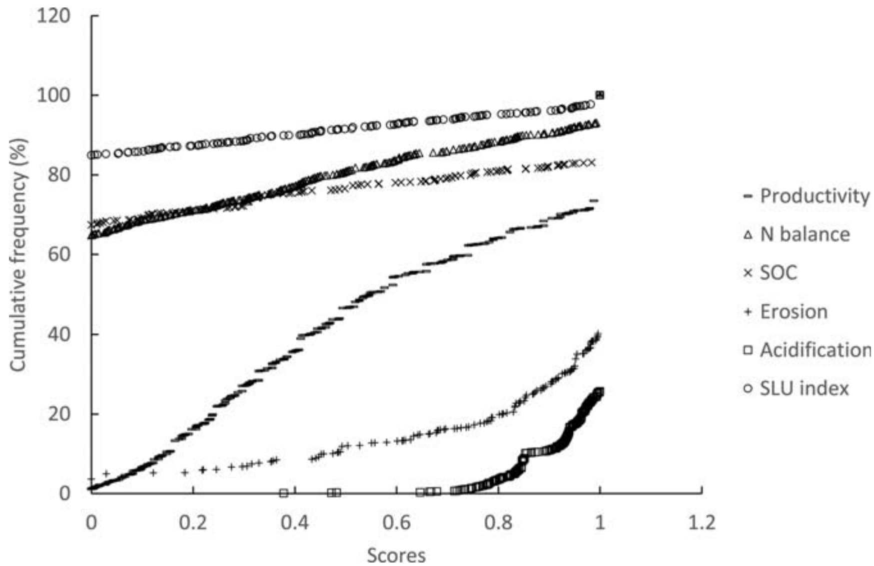

Figure 4. Frequency distribution of indicator scores related to crop productivity, partial N balance, soil organic C, soil pH, soil losses through erosion and consequent Sustainable Land Use (SLU) index in the Tanzania case study.

6. Discussion

The application of the framework to the case-study in Tanzania, which included diverse biophysical conditions (topography, rainfall, soils) and diverse management practices (nutrient sources, SCP, crop residue management) revealed that the framework detected differences in SLU indicators as a result of management practices. The result of the case-study showed the management practices that should be promoted at large-scale (community level) within the study area, because the indicator gap occurred in most plots and across the districts. For instance, the predominant negative N balances suggests that nutrient application should be promoted at large-scale in the area. Negative N balances, comparable to those in the case-study, have repeatedly been reported in studies in the region (Kihara et al., 2015; Tully et al., 2015; Vitousek et al., 2009). In the study in western Kenya, Tully et al. (2015) reported negative balances ranging from -1 to -112 kg N ha^{-1} in 17 of the 24 farms (71%) that they monitored, while positive balances of 13–93 kg N ha^{-1} were also observed in other farms. Kihara et al. (2015) reported negative N balances in 74% of the 117 fields that they studied in Tanzania. Given that fertilizers are viewed as the main entry point for agriculture intensification and for reversing the nutrient depletion trend in the region, efforts towards increasing the use of fertilizers should be accentuated (Jayne et al., 2019). In the case-study, 59% of plots received fertilizers but at a low rate (average N application rate was 38 kg N ha^{-1}), similar to the average rate of 32 kg N ha^{-1} in Tanzania reported by Holden (2018) based on Living Standard Measurement Survey. Increasing farmers' access to fertilizers has been a focus of various national and sub-regional initiatives in SSA towards achieving the SDG 2 on eliminating hunger (Jayne et al., 2018). This increase in nutrient additions should be accompanied by managements practices that can maximize the nutrient use efficiency and minimize unintended losses to the environment.

The low percentage (17%) of plots that attained the sustainability level for $C_{applied}$ indicates a general insufficient use of organic inputs, the main contributor to SOC build-up (Fujisaki et al., 2018; Lal et al., 2007). An increase in the use of fertilizers (as suggested above) can enhance the $C_{applied}$, due to increased crop yields including crop residues, provided the residues are not removed from the plot (Hijbeek et al., 2019; Lal et al., 2007). Increasing the use of organic inputs should be included in the recommended management practices at community level. Efforts to increase the use of organic inputs in SSA have been constrained by the limited availability of organic sources under smallholder farming conditions, competitive use and the lack of incentives for technologies to produce them (Castellanos-

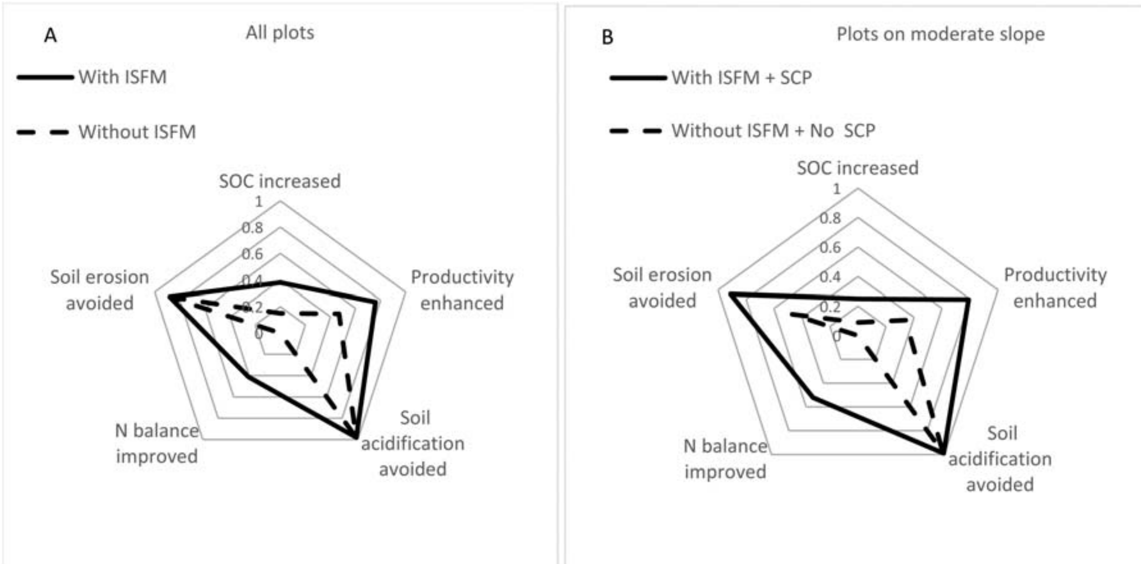

Figure 5. Indicator scores in the Tanzania case study as influenced by integrated soil fertility management (ISFM) and soil conservation practices (SCP). A: Considering all plots included in the case study, B: Considering only plots positioned on a moderate slope in the case study.

Navarre et al., 2015; Coulibaly et al., 2017; Place & Dewees, 1999). The integrated soil fertility management (ISFM) approach reconciles the limited availability of both fertilizers and organic inputs by recommending their co-application and with it also improved germplasms to maximize the use efficiency of applied nutrients (Vanlauwe et al., 2010). In the case study, the proportion of plots under which the ISFM package was applied was very low (6%) but resulted in better outcomes for indicators than in plots where no ISFM was applied (Figures 3C and 5A).

The estimated soil losses in the case study are comparable with those reported in various studies using RUSLE model in the East Africa region, although assessment in those studies used a catchment or landscape approach. Haregeweyn et al. (2017) reported average soil losses of 27.5 ton ha^{-1} yr^{-1} (range 0–200 ton ha^{-1} yr^{-1}) in the Upper Blue Nile river basin in Ethiopia, with severe erosion attributed to high population densities. Tamene and Le (2015) estimated soil losses in the whole Nile river catchment (including Burundi, Rwanda, Tanzania, Uganda, Kenya, Ethiopia) to average 6 ton ha^{-1} yr^{-1} in the white Nile and 85 ton ha^{-1} yr^{-1} for the Blue Nile. Karamage et al. (2016b) estimated average soil losses of 15 ton ha^{-1} yr^{-1} in land classified as suitable for agriculture in the Nyabarongo river catchment, in Rwanda. As in the case study, all these studies observed a wide range of soil losses, with areas on steep slopes exhibiting the highest values, and skewing the average soil erosion estimates.

Soil conservation practices such as mulching, hedges, and contour ploughing, have been shown to reduce the soil losses and could be promoted to farmers (Danga & Wakindiki, 2020). Haregeweyn et al. (2017) predicted a reduction of the mean soil losses observed in 2016 from 27.5 to 13.5 ton ha^{-1} $year^{-1}$ by 2025 if soil conservation practices are used. In the case-study, plots where soil conservation practices were applied had lower soil losses (better scores) compared to plots where those practices were absent (Figure 5B).

Most plots (>70%) in the study area have acceptable soil pH for the sustainability objective, suggesting that acidification may not be an issue and that practices to manage it could target the individual plots where a subtle gap was identified.

Addressing most of the above would be beneficial for crop productivity, which was predominantly low in the study area, given that a crop productivity score of 1 was only reached in 26% plots. The mean maize productivity in non-ISFM plots (~1 ton ha^{-1}) is similar to yields reported in various studies in the region where no inputs are applied (Kihara et al., 2015; Tittonel & Giller, 2013).

While approaches used for indicator estimations are based on well documented information, data needed for parameters used in some indicators are scanty in SSA. We have suggested potential data sources for situations where data needed to quantify specific metrics are not available. Notwithstanding availability of regional data, context specific data is more appropriate and recommended. For example, BNF and harvest indices (HI) can vary between crop varieties. Likewise, soil specific measurements are needed until suggested secondary data layers, such as those of the SoilGrids (https://soilgrids.org) become very specific. It is expected that data availability will increase as tools and data sources evolve, providing an opportunity to generate more context specific data for the framework. Therefore, it should be noted that this framework is not intended to generate exact values of indicators, but rather estimates that are useful for monitoring changes due to agricultural management practices. Such estimates are also useful to determine whether an agricultural practice has a potential to contribute to SLU by assessing its impact on multiple indicators. Often, management practices are developed and evaluated with focus on optimizing one indicator, whereas the practices generally affect multiple indicators.

Data generated from a continuous assessment (with this framework) can be potentially used to develop predictions regarding trajectory and timeline for reaching sustainability based on a set of management practices. This is useful to understand the long-term effects of practices, specifically for indicators with thresholds which allow some level of damage (e.g. erosion). There is a need to apply the framework on plots where conventional methods for quantifying indicators are implemented so to generate comparative data to validate the indirect assessment for various cropping systems. This can be achieved by a first identification of research projects/initiatives focusing on specific indicators, followed by an interaction with the implementers to collect information needed for the indirect assessment.

We have limited the assessment to the plot level because agricultural practices are directly applied to plot. However, practices applied to a plot can have effects on neighbouring plots within a farm. For example, the use (or the lack) of soil conservation measures on a plot would reduce (or increase) erosion on another plot. The application of the framework for farm level SLU assessment would require consideration of the interaction effects of practices between plots. Often, farm level assessments are achieved by aggregating plot level assessments without consideration of the between plot effects, mainly due to the lack of suitable methodologies to account for such effects (Marinus et al., 2018).

The framework is developed for use primarily in the African smallholder farming context, where deployment of conventional methods for quantifying indicator metrics is logistically constrained, whereas agricultural management practices can rapidly impact the already fragile land. However, it can be applied elsewhere with adjustment to adapt some parameters to the local/regional context. This applies particularly to the erosion indicator where the estimation of R is based on the MFI equation recommended for Africa. Adjustment of R estimation to the model adapted to the conditions would be needed.

The framework is currently limited to five indicators. However, it can be updated to include other SLU indicators that are relevant for the specific context if their relationship with management practices is quantitatively established. The process for insertion of the new indicators will follow the same procedure as described, including defining critical thresholds.

Target users of this framework are the research community. Firstly, researchers can use the framework to check if technologies or interventions that they seek to promote are suitable to contribute to the SLU objective. With the undisputable need for sustainable intensification in SSA, it is imperative to develop and promote technologies that meet this objective. The framework offers an opportunity to demonstrate the contributions of technologies and interventions to SLU, as an important component of SI, by assessing their impacts on multiple indicators, instead of a focus on one indicator as often the case. Secondly, the research community can use the framework as a tool to guide farmers and agricultural service providers on effective practices to turn land into sustainable use. Typically, farmers apply agricultural practices with minimal understanding of their broader implication on the sustainability of the agroecosystem. Therefore, researchers can work with agricultural extension service providers, to collect information on agricultural practices used by farmers on their land, apply them into the framework, and provide feedback with advices on suitable practices to advance the sustainability objective. By adjusting to these practices, farmers will improve and sustain the productivity of their farm and contribute to the protection of natural resources.

7. Conclusions

We have developed a framework that can be rapidly applied (i) to assess the gap in sustainable use of a land, (ii) to identify indicators which require most attention to be addressed, (iii) to advise farmers on appropriate management practices to turn land into sustainable use, and (iv) to monitor progress in land use due to changes in management practices; as shown for the case-study conducted in Tanzania. Indicator values obtained in the case-study are comparable to values reported in studies conducted in the region, though better localized data are needed for various parameters used in the estimation of the indicators. The application of the framework can be useful for initiatives that are focused on advancing the sustainable intensification of smallholder agriculture in SSA, by ensuring that technologies and management practices aiming at increasing food production are viable to reverse the trend of land degradation.

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The authors declare no conflicting interest that they have no affiliations with or involvement in any organization or entity with any financial or non-financial interest in the subject matter or materials discussed in this manuscript.

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