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6. Land Use Impacts on Soil Quality

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6.1. Scope

Soils are the loose upper layer of the Earth's surface, composed of '*weathered mineral materials, organic material, air and water*' (FAO 2018). According to a common definition from soil scientists, *"soil quality is the fitness of a specific kind of soil to function within its surroundings, support plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation"* (Karlen, Mausbach et al. 1997). This definition emphasises both inherent properties of soil ("*a specific kind of soil*") and dynamic interactive processes (Larson and Pierce 1991), and links soil quality to its functions, which contribute to ecosystem services.

Soils contribute to ecosystem services including: i) provisioning (e.g., fresh water), ii) regulating (e.g., climate regulation), iii) cultural (e.g., recreation) and iv) supporting services (e.g., primary production) (World Resources Institute 2005; Dominati, Patterson et al. 2010; Adhikari and Hartemink 2016; Cowie, Orr et al. 2018). The central role of soils for ecosystem services justifies that they are addressed specifically across several Sustainable Development Goals (SDGs), including 2.4 (sustainable food production systems) and 15.3 (striving to achieve a land degradation-neutral world). Due to the complex spatial and temporal characterisation of soil functions and properties across multiple Earth's spheres (lithosphere, biosphere, atmosphere, hydrosphere) and scales, modelling of soil processes and their associated services is a challenging task (Adhikari and Hartemink 2016).

Soil functions are determined by pedoclimatic variables such as soil texture, soil organic matter, rainfall, temperature, and related biological parameters. Assessing the effects of human interventions on soil quality requires a geographic scale that is sensitive to these variables and is relative to the optimum soil quality possible within a given context.

Land use and land use change (LULUC) are key human stressors that affect soil quality, e.g., by modifying physical, chemical, and biological properties of soil through agriculture and forestry, by altering the rate of removal of soil, and/or sealing it through infrastructure. Other significant impacts on soil quality can be caused by the presence and/or accumulation of contaminants in soil, leading to acidification or toxic impacts. The potential impacts of human interventions on soil quality through LULUC and the associated management practices make the inclusion of a soil quality indicator essential for many life cycle assessment (LCA) studies of product systems that transform or occupy land. This chapter focuses exclusively on the impacts from LULUC on soil quality and does not address toxicity or eutrophication, which are dealt with in other chapters.

6.2. Review of approaches and indicators

This section reviews existing life cycle impact assessment (LCIA) methods with available characterisation factors that are relevant for soil quality and that were considered for recommendation. Current LCIA models do not provide a harmonised comprehensive assessment of soil quality. They focus on various indicators covering various physical, chemical, and biological properties. The most common models address soil organic carbon, soil erosion, and biological productivity. Only one model, the land use indicator value calculation in life cycle assessment (LANCA®) model (Bos, Horn et al. 2016), combining several approaches, also includes groundwater regeneration, mechanical filtration, and water infiltration capacity. We detail hereafter the first three most-encountered indicators. A more comprehensive description of the methods and models is given by Vidal Legaz, Maia De Souza et al. (2017), including key methodological elements and original sources.

6.2.1 Soil organic carbon

Change in soil organic carbon (SOC), usually in kgC/m^2, has been proposed to be used in LCIA by several authors (Mattsson, Cederberg et al. 1990; Cowell 1998; Baitz, Kreißig et al. 1999; Milà i Canals and Polo 2003; Milà i Canals, Romanya et al. 2007; Brandão and Milà i Canals 2013) as a good indicator for soil quality. It is causally associated with other important indicators including: soil fertility and biotic production; carbon and nutrient cycling; and water infiltration and erosion protection (see Figure 6.1). The ultimate effects of SOC change on final ecosystem services (e.g., those delivered to ultimate human beneficiaries) require further modelling to consider the change in biotic production and the fraction of this biotic production that provides benefits to humans (e.g., climate change mitigation, provision of biomass for food, fibre, and feed).

The change in SOC is measured relative to a reference state which Brandão and Milà i Canals (2013) term potential natural vegetation (PNV). The SOC values have broad global coverage and are geographically differentiated based on climate regions. As for the coverage of land and management practices, Brandão and Milà i Canals (2013) include different SOC characterisation factors values for land use intensity (low input, medium input, high input without manure, high input with manure) and tillage practice (full tillage, reduced tillage, no tillage) based on the parameters used in the Intergovernmental Panel on Climate Change (IPCC) calculations for total soil carbon change.

6.2.2 Biomass production

Biomass production is correlated to a change in SOC, itself influenced by the combination of other factors including soil type, climate region, land cover, and management practices. Indeed, SOC reflects the balance between inputs of organic matter derived from biotic production and the turnover related to soil biological activities. Hence, changes in biotic production leads to a new equilibrium SOC level. As soil quality determines productive potential, and therefore biomass production, change in biomass production as indicated by change in SOC is relevant to soil quality.

Impact on biomass production is proposed in a number of LCIA models (Núñez, Antón et al. 2013; Alvarenga, Erb et al. 2015; Bos, Horn et al. 2016), although different indicators and pathways are considered in each model. Net primary production depletion (NPPD) was proposed by Núñez, Antón et al. (2013) in which soil loss through erosion is linked to a loss of biomass production (as well as damage to natural resources). Human appropriation of NPP (HANPP) was proposed by Alvarenga, Erb et al. (2015), which measures the NPP consumed by humans and, therefore, not available for ecosystems, by looking at the difference between NPP of potential natural vegetation and the current land use. Biotic production loss potential (BPLP) proposed by Bos, Horn et al. (2016) is obtained by the difference in biomass production between the current and the reference land use. Where a PNV reference state is used for BPLP it would represent the same outcome as HANPP, with both measuring the difference in biomass production between the modified land use and a PNV baseline.

In Núñez et al. (2013) and in Bos, Horn et al. (2016) the value of NPP is dependent on climate, soil properties, and the sealing factor.

6.2.3 Erosion

Erosion can be modelled in LCIA as a result of LULUC as is done by Bos, Horn et al. (2016) but there are methods that add erosion as a life cycle inventory (LCI) elementary flow, as is done by Núñez, Antón et al. (2013). In both of these methods, water erosion potential is calculated using the revised universal soil loss equation (RUSLE) (Renard, Foster et al. 1991), which depends on soil characteristics, rainfall, slope, land cover, and management activities (Bos, Horn et al. 2016). In this report the LCIA pathway is examined as it is better suited to existing LCI datasets.

6.2.4 Other impact categories

In addition to the indicators presented above (SOC, biotic production, and erosion), models have also been proposed for groundwater regeneration, water infiltration capacity, and physicochemical filtration reduction.

Bos et al. (2016) describes how different types of land use can contribute to soil sealing, which can affect surface water flow and evapotranspiration, and subsequently the rate of groundwater regeneration.

Bos et al. (2016) is the only LCIA model that accounts for the impact of soil sealing on water infiltration capacity. Infiltration reduction potential (IRP), an indicator representing a loss of soil mechanical filtration capacity, is affected by soil characterised permeability [cm/(d*m^2)], which depends on soil, land use type, distribution of pores, and depth to groundwater table (Beck, Bos et al. 2010; Bos, Horn et al. 2016).

Soil regulates water flow and the transport and storage of other substances that can affect water quality. Changes in organic matter affect its capacity to store substances such as nitrogen compounds that can affect water quality. Bos, Horn et al. (2016) use cation exchange capacity (CEC) as the corresponding impact indicator. Soil organic matter, of which SOC is the major component, accounts for the largest share of CEC in mineral soils (50-90%) and is thus a key indicator for the filtration and buffering capacities of soils (Brady and Weil 1999).

6.2.5 Summary of approaches and indicators

In conclusion, there are currently a limited number of LCIA models covering a number of impact pathways addressing soil quality issues. However, there is still no comprehensive approach to soil quality assessment. Soil processes are complex, and the impacts of LU are wide-ranging. More research work is needed in order to develop a model assessing, if not all indicators, at least a comprehensive set of soil quality-mediated impacts of LU on ecosystem services and other damage categories. New models are also needed for connecting the impacts from land use occupation and transformation to the damages endpoints defined by Verones, Henderson et al. (2016). Moreover, only a few of these models allow for deriving characterisation factors with both global coverage and regionally specific declinations.

6.2.6 Reference states

Since evaluating the environmental effects of land use is always in comparison to a reference situation, this land use reference state needs to be clearly defined. Saad, Koellner et al. (2013) propose the use of potential natural vegetation (PNV) at the scale of terrestrial ecoregions based on Olson *et al.* (2001) as the reference state. Bos, Horn et al. (2016) defined their reference as the largest natural biome (in terms of surface area) in each country, but this created anomalies especially in countries with large low-productivity land or land not managed for agricultural or forest production such as Australia, with desert being the reference state. In an update to the factors in 2019, the reference state was calculated as a weighted average of the values of ecosystem quality for all the types of PNV that can be found in a country according to the global map of ecological zones provided by FAO (2012). The weighted average was calculated considering the area share of each ecological zones in a country and excluding, for agricultural and forest-related land use types, the following ecological zones: "boreal tundra woodland," "polar," "subtropical desert," "temperate desert," and "tropical desert" (De Laurentiis, Secchi et al. 2019). Although Saad et al. (2013) also used the framework developed in the LANCA model, they propose that the reference state be based on PNV using Holdridge life zone level (Holdridge, 1947 #2047) (a combination of climatic conditions and vegetation cover that provides simpler classification–only 38 life zones globally, compared with 867 ecoregions). This potentially provides a more comparable baseline than used in LANCA and is consistent with the baseline chosen for the biodiversity model provisionally recommended for land use impacts on biodiversity at the last Pellston Workshop (Chaudhary, Verones et al. 2015), which was the natural or close to natural vegetation habitat per ecoregion (Frischknecht and Jolliet 2016).

6.3. Process and criteria applied and process to select the indicator(s)

For the purpose of incorporating soil quality impacts within LCA, ideally the choice of the indicators should comply with the following criteria:

- Soil quality should be represented by a minimum number of indicators, in order to avoid the multiplication of recommended indicators, with causal links to the main soil functions to enable efficient interpretation of impacts;
- The indicator should be compatible with existing land use LCI flows, i.e., related to land occupation and transformation elementary flows, but may also recommend additional elementary flows;
- The indicator should be applicable globally, to all types of land use, for both background and foreground processes.

One approach to derive a metric for soil quality is through a soil quality index (SQI).

The Swiss agricultural life cycle assessment (SALCA) model (Nemecek, von Richthofen et al. 2008) combines several indicators using a mechanistic process-based composite model into an SQI, but was not considered for inclusion in this assessment, as it was developed specifically for the Swiss context and relies mostly on expert knowledge and detailed primary data. The detailed data requirements make this model incompatible with the global scope of LCIA.

A type of SQI was developed by the European Commission's Joint Research Centre (JRC) that included aggregating four of five indicators from the LANCA model (Bos, Horn et al. 2016). However, the weighting system was viewed as subjective, as all indicators were given the same weighting in the index. As physicochemical filtration and mechanical filtration indicators presented a unitary correlation coefficient, only the latter was included in the aggregation to avoid

accounting for potentially redundant information (De Laurentiis, Secchi et al. 2019).

Several authors agreed that a contribution based on observed correlations using multi-parametric statistics could be used to link soil quality attributes to a one-dimensional SQI (Andrews, Karlen et al. 2002; Velásquez, Lavelle et al. 2007; Obriot, Stauffer et al. 2016). However, such an approach is not currently available, and failing this, there is no consensus on alternative approaches to calculate a soil quality index (Andrews, Karlen et al. 2004; de Paul Obade and Lal 2016; Obriot, Stauffer et al. 2016).

Given the limitations of existing LCIA models described above, soil organic carbon (SOC) remains the only available indicator that is both comprehensively linked to several soil quality functions and is applicable within the LCIA framework. SOC is a soil property that mediates many cause-effect links between soil properties and soil functioning (Dominati, Patterson et al. 2010; Cowie, Orr et al. 2018). In particular, SOC is to some extent an implicit indicator of the amount of soil biota present in soil. SOC is positively correlated with the four key soil functions as defined by Kibblewhite, Ritz et al. (2008): carbon transformations; nutrient cycling; soil structure maintenance; and the biological population regulation of soil fauna. As summarised by FAO[13] "SOC transcends all chemical, physical, and biological soil indicators and has the most widely recognized influence on soil quality as it is linked to all soil functions." We hence recommend using change in SOC, in kg C, as a midpoint indicator for soil quality impacts from LULUC.

It is recognised that SOC does not represent all aspects of soil quality. Erosion, chemical pollution and salinization are processes that have a weak correlation with SOC (Milà i Canals and Polo 2003; Milà i Canals, Romanya et al. 2007). Since soil loss is considered a critical soil threat (Yang, Kanae et al. 2003), we also therefore recommend quantifying soil loss from erosion using the LANCA model (Bos, Horn et al. 2016), initially as a midpoint, but ultimately with a view to include it within the socio-economic assets damage category. Chemical pollution of soils is covered through ecotoxicity and acidification indicators, while salinization is not currently considered.

6.4. Description of the impact pathway and indicators selected

Figure 6.1 presents the impact pathway linking LULUC to the damage categories via processes that impact soil properties, soil functioning, and ecosystem services. The life cycle inventory is based on land occupation and land transformation under different types of land cover and land management. These occupations and transformations can include processes (e.g., sealing, compaction, etc.) that have direct impacts on soil properties (e.g., SOC content, soil structure, soil

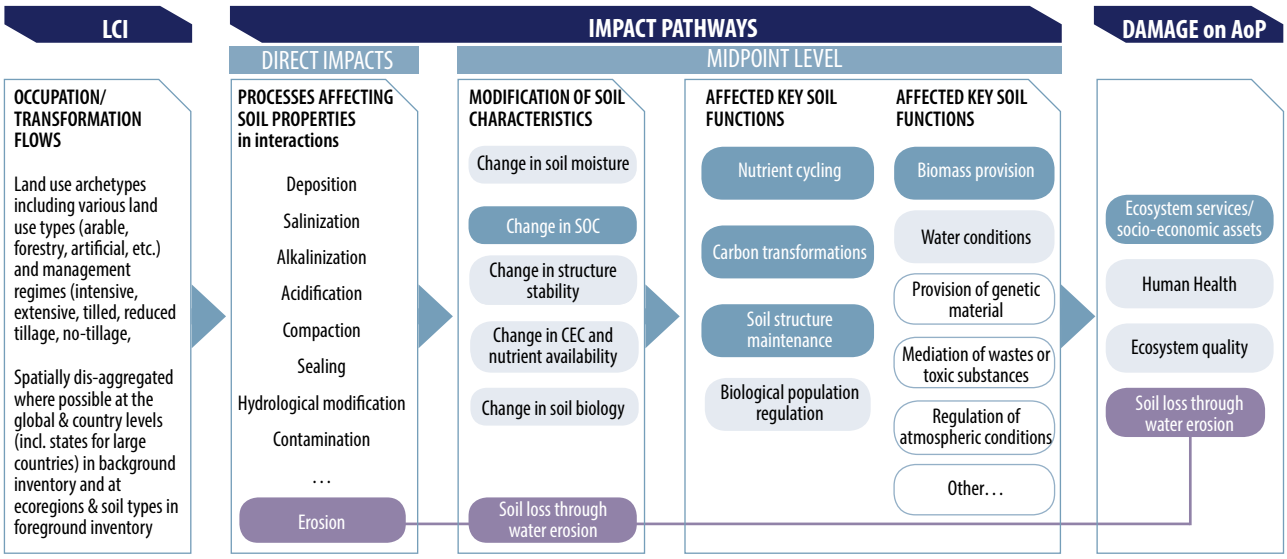

Figure 6.1. Impact pathway of land use impact on soil quality and soil loss through water erosion

13 http://www.fao.org/soils-portal/soil-degradation-restoration/global-soil-health-indicators-and-assessment/en/

moisture) and can affect soil functions (e.g., nutrient cycling) and the ability of soils to provide ecosystem services (e.g., biomass provision).

Figure 6.1 shows the two selected indicators with the cells highlighted in dark blue being the impact pathway for change in SOC and cells highlighted in red is the pathway for soil loss through water erosion. While three soil functions are identified in the change in SOC pathway, many soil functions and ecosystem services are connected and are shown in lighter shades of blue to represent softer links to the SOC pathway. Similarly, for erosion the impact pathway is connected to natural resources but may also have links to other damage categories such as socio-economic assets. For simplicity, we present just a few key ecosystem services from the Common International Classification of Ecosystem Services (CICES)[14]. Additional ecosystem services may be affected by other changes in soil quality due to LULUC that are not captured in this figure.

6.5. Model, method and specific issues addressed

6.5.1 Calculation of land occupation and transformation impacts

The two types of land use interventions as described by Koellner, De Baan et al. (2013) are land transformation and land occupation. In the context of our soil quality indicators, land occupation is quantified by the area of land and the time for which it is occupied in units of m^2•year. The impact of occupation is the difference between the quality of the land in the occupied state and the reference state as shown in Equation 1. The reference state in the selected methods is potential natural vegetation.

Equation 1

$$\text{Occupation impact} = (A_{FU} * t_{FU}) * (Q_{Ref} - Q_{LU})$$

Where Occupation impact is in kg SOC deficit•year or kg soil lost

A_{FU} is the area of the functional unit in m^2,

t_{FU} is the time of occupation for the functional unit in years,

Q_{Ref} is the quality of the land in the reference state in kg C or kg soil loss/ year, and

Q_{LU} is the quality of the land in the occupied state in kg C or kg soil loss/ year.

Transformation is measured in m^2 as the act of transformation is taken to occur over a short period and in LCA is assumed to be instantaneous. Physical damage to land during the process of transformation is not quantified in LCA[15]. What is quantified are the impacts of occupying the land during the time it takes to return it to the quality under the prior land use, with the time taken referred to as the regeneration time. Transformation effects are calculated as the difference in land quality multiplied by half the regeneration time as shown in Equation 2.

Equation 2

$$\text{Transformation impact} = A_{FU} * (Q_{LU2} - Q_{LU1}) * t_{reg} / 2$$

Where Transformation impact is in kg SOC deficit•year or kg soil lost

A_{FU} is the area of land transformation in m^2,

Q_{LU2} is the quality of the land after transformation in kg C or kg soil loss/year,

Q_{LU1} is the quality of land prior to transformation kg C or kg soil loss/year, and

t_{reg} is the regeneration time to achieve the quality of the prior land use in years.

Life cycle inventories generally include paired flows for "land transformation from," and "land transformation to" so the quality change is measured from the reference state. However, when the two are combined the reference state cancels out of the equation.

6.5.2 Soil organic carbon (SOC) deficit potential

SOC deficit potential has been defined as the change in soil organic carbon (ΔSOC) over a period of time relative to a PNV reference state. ΔSOC is recommended as a midpoint impact indicator with further investigation required to link this to related damage category.

We retain the model presented by Brandão and Milà i Canals (2013) for calculating characterisation factors (CFs) for SOC deficit potential. The proposed CFs for land occupation are defined as the ΔSOC between the reference land use and the current land use over

14 https://cices.eu/cices-structure/

15 For example, when clearing natural vegetation for use as cropping land the erosion from tree removal prior to establishment of the crop is typically not quantified.

the occupation time (kg C * m^{-2}). Multiplying the land occupation CF by the land occupation inventory flow (m^2 • year) results in a ΔSOC in units of kg C • year, which represents the time integrated ΔSOC between the reference land use and the current land use (SOC deficit potential).

Characterisation factors for land transformation represent the time-integrated ΔSOC during the regeneration time between the previous land use and the new land use in kg SOC (kg SOC * m^{-2} * yr). Multiplying the transformation characterisation factor by the inventory flow for land transformation (m^2) results in a transformation impact of ΔSOC in units of kg SOC•year (SOC deficit potential). Only national average characterisation factors have been developed based on potential natural vegetation as the reference land use.

CFs proposed in Brandão & Milà i Canals (2013) are based on default SOC data for climate regions and soil types and under different land use and management conditions reported by the IPCC (Eggleston et al. 2006). These IPCC estimates are based on soil data from the National Soil Characterization Database (USDA 1994), the World Inventory of Soil Emission Potential Database (International Soil Reference and Information Centre), and data on SOC compiled by (Bernoux, da Conceição Santana Carvalho et al. 2002). For the purpose of greenhouse gas (GHG) accounting, IPCC 'default data' is applicable to the simplified tier 1 accounting methods, whereas tier 2 and tier 3 approaches are recommended when more specific data are available. These more detailed approaches could be used to improve the SOC deficit model.

6.5.3 Erosion potential

Soil erosion induced by movement of water is estimated using the revised universal soil loss equation (RUSLE), based on the approach by Bos et al. (2016) with revisions (Horn and Maier 2018) to the reference state mentioned in Section 6.2.6 and the inclusion of regeneration times for land transformations from Brandão and Milà i Canals (2013). These regeneration times were added for compatibility with the approach in the SOC deficit potential indicator and LCIA of land use approach recommended by UNEP-SETAC (Koellner, De Baan et al. 2013). Characterisation factors for soil erosion from water include land occupation effects in kg soil m^{-2} $year^{-1}$ and land transformation in kg soil m^{-2}. Multiplying the occupation and transformation characterisation factors by the inventory flow for land occupation (m^2•year) and land transformation (m^2) results in a occupation and transformation impact in kg soil loss (erosion potential).

LANCA (Bos, Horn et al. 2016) also suggests to consider some land transformations with no possible regeneration as permanent transformation, which is not considered in the presented approach. To avoid leaving these transformations out of LCA, the regeneration time from the supplementary data set provided in Brandão & Milà i Canals (2013) of 85 years has been applied for transformation to completely denuded areas such as construction and mining sites.

6.6. Characterisation factors

6.6.1 SOC deficit potential

CFs for SOC deficit potential are available for nine land use types in ten climate region levels and for six soil types, plus one set for global-default values based on the weighted average of the ten climate regions. In addition, several intensities for several of the land use types are provided (namely four different intensities for each of full tillage, reduced tillage, and no tillage agricultural land; three levels of degradation and two intensities for grasslands).

As no country specific CFs were provided in Brandão & Milà i Canals (2013), the following procedure is applied to provide these CFs: for each country considered, the geographical distribution of climate types (Joint Research Centre 2018) in that country are used to calculate country-specific characterisation factors. Then country-specific CFs are calculated as an area-weighted average of the CFs provided for the different climate regions. When aggregating, only areas where a certain land use activity can take place are considered, excluding deserts and permanent snow-covered areas from the aggregation.

Table 6.1 shows the global factors from Brandão & Milà i Canals (2013), as well as derived factors for one country (China) using the climate regions calculation.

6.6.2 Erosion potential

For the quantification of erosion potential from land use activities, the set of characterisation factors proposed by Bos et al. (2016) are recommended

Table 6.1. Occupation and transformation CFs for soil organic carbon deficit potential (for global average and 1 country example [China]) for different land use types

Land use	Land use sub-category	World (occup.) tC.yr ha^{-1}yr^{-1}	China (occup.) tC.yr ha^{-1}yr^{-1}	World (trans.)	Transformation avg global
Long-term cultivated	Unspecified	21	16.0	205	160.0
Long-term cultivated full tillage	Unspecified	21	16.0	205	160.0
	Low input	23	18.7	231	187.4
	Medium input	21	16.0	205	160.0
	High input without manure	17	12.9	175	129.1
	High input with manure	5	-1.4	50	-13.8
Long-term cultivated Reduced tillage	Unspecified	18	13.8	176	138.2
	Low input	20	16.7	203	167.2
	Medium input	18	13.8	176	138.2
	High input without manure	14	10.5	142	105.3
	High input with manure	1	-4.5	8	-44.8
Long-term cultivated No tillage	Unspecified	15	10.6	148	105.5
	Low input	18	13.7	177	136.5
	Medium input	15	10.6	148	105.5
	High input without manure	11	7.0	112	70.4
	High input with manure	-3	-9.1	-31	-90.6
Permanent grassland	Permanent grassland	0	0.0	0	0.0
	Nominally managed (non-degraded)	0	0.0	0	0.0
	Moderately degraded	2	2.9	24	28.8
	Severely degraded	17	17.8	175	178.0
	Improved grassland - medium land management	0	0.0	0	0.0
	Improved grassland - high land management	-6	-6.5	-64	-65.2
Paddy rice		-6	-5.9	-58	-59.3
Perennial/Tree Crop		0	0.0	0	0.0
Set-aside (< 20 yrs)		8	7.4	80	73.6
Sealed Land		58	59.3	2465	2926.9
Forest		0	0	0	0

with the modifications to reference states discussed in Section 6.2.6, and inclusion of regeneration times discussed in Section 6.5.2, which have been implemented in the latest release (Horn and Maier 2018). CFs are provided by Bos et al. (2016) at both the global and country scale for a list of 58 land use types. A selection of the CFs calculated, based on the requirements of the rice case study to follow, are shown in Table 6.2.

6.6.3 Summary of proposed CFs

In summary, the new set of CFs was developed for both indicators. For the CFs for SOC, this was done by providing aggregated CFs at country level using climate data for each country and applying the climate-based CFs based on an areas weighted average.

The set of CFs to be used in the calculation of impacts on erosion potential were obtained by modifying the CFs provided in Horn and Maier (2018) and including the regeneration time in the calculation of transformation CFs.

In order to be consistent with the assumptions used in the calculation of the SOC CFs, regeneration times were taken from Brandão & Milà i Canals (2013) equal to 20 years for biotic land uses and 85 years for sealed land.

6.7. Rice case study application

A rice LCA case study was developed based on Frischknecht, Fantke et al. (2016) to illustrate the practical application of the proposed set of CFs and to identify needs for future development to improve their applicability in LCA. The case study includes three

Table 6.2. Selection of occupation and transformation CFs for erosion potential for global and 1 country example (China)

	World (occup.) kg soil $ha^{-1}yr^{-1}$	China (occup.) kg soil $ha^{-1}yr^{-1}$	World (trans.) kg soil ha^{-1}	China (trans.) kg soil ha^{-1}
Unspecified	-0.708	-1.060	-30.100	-45.200
Unspecified, natural	-0.661	-1.050	-6.610	-10.500
Forest, natural	-0.013	-0.005	-0.134	-0.052
Forest, secondary	0.007	0.001	0.072	0.013
Wetlands	-0.723	-1.070	-7.230	-10.700
Shrub land	-0.640	-1.040	-6.400	-10.400
Grassland/pasture/meadow	0.048	0.014	0.484	0.142
Grassland	0.048	0.014	0.484	0.142
Pasture/meadow, extensive	0.028	0.008	0.278	0.077
Agriculture	6.130	1.910	61.300	19.100
Arable	8.200	2.560	82.000	25.600
Arable, fallow	10.300	3.200	103.000	32.000
Arable, extensive	7.160	2.240	71.600	22.400
Arable, intensive	9.230	2.880	92.300	28.800
Arable, flooded crops	-0.034	-0.012	-0.341	-0.116
Arable, greenhouse	-0.040	-0.014	-0.403	-0.135
Permanent crops, extensive	7.160	2.240	71.600	22.400
Agriculture, mosaic	6.130	1.910	61.300	19.100
Urban	-0.708	-1.060	-30.100	-45.200
Industrial area	-0.712	-1.070	-30.300	-45.300
Construction site	13.700	3.440	583.000	146.000
Traffic area, rail/road embankment	-0.578	-1.020	-24.600	-43.500
Bare area	19.900	5.370	199.000	53.700

scenarios for rice production and use with one being rice grown in the U.S. and consumed in Switzerland, the second being rice grown and consumed in China, and the third being rice grown and consumed in India. Table 6.3 shows the inventory data for land occupation (m^2y) differentiated according to the land use classes used by Brandão & Milà i Canals (2013), and between foreground and background activities.

Table 6.3. Land occupation life cycle inventory results (cumulative land occupation) per land use classes [m^2y]

	USA - Switzerland	China	India
Annual crop (foreground)	1.40	1.46	2.69
Annual crop (background)	0.00	0.00	0.00
Perennial/tree crop (foreground)	0.00	0.00	0.46
Perennial/tree crop (background)	0.13	0.11	0.10
Sealed land (foreground)	0.01	0.01	0.00
Sealed land (background)	0.10	0.01	0.01
TOTAL	1.64	1.59	3.26

The CFs are provided aggregated at country level calculated from the original climate region CFs using areas of each climate type in each country. Information available in the rice case study was at the resolution of specific countries in which foreground activities would take place (i.e., the U.S., Switzerland, India, and China), and so country-specific characterisation factors were used for all foreground activities. The case study was also conducted using the global-default CFs, to assess how this would affect the results.

Case study results and discussion

Figure 6.2 shows the contribution analysis results for SOC deficit potential and erosion potential of the rice case study based on country-specific CFs for foreground activities. The agricultural phase (production) has the largest contribution for both indicators. For China, rice distribution also has noticeable impacts on SOC deficit potential. The SOC deficit potential from distribution is due to the high CF assigned to occupation as sealed land. Conversely, for erosion potential a negative impact (i.e., a benefit) is assigned to the distribution phase due to the negative CFs for occupation of artificial areas. Since the CF for

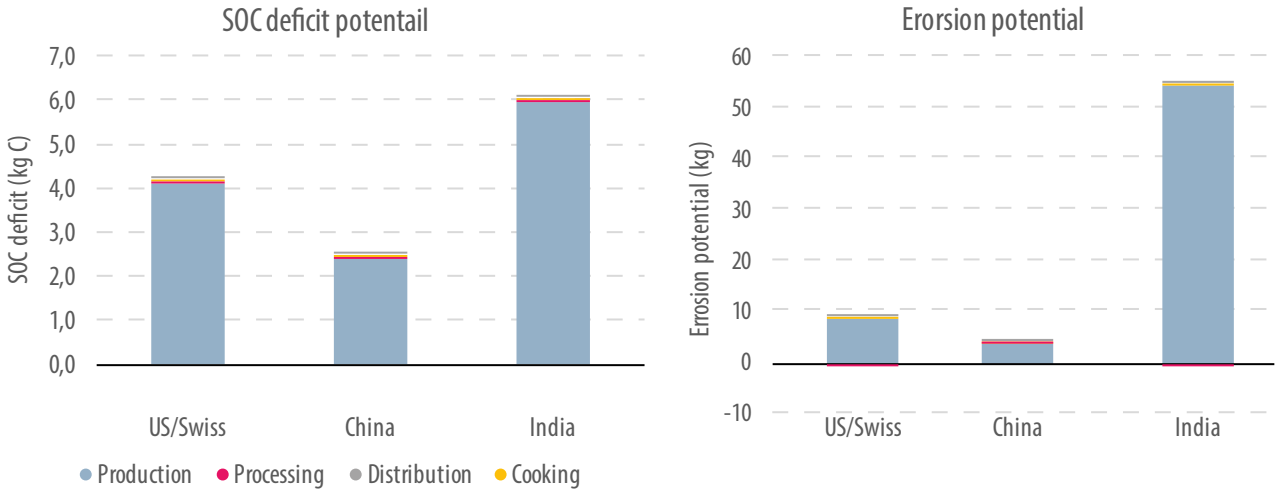

Figure 6.2. Contribution analysis for rice production: results for SOC deficit potential and erosion potential per kg rice cooked.

erosion potential is limited to water erosion, additional soil loss processes like wind erosion and soil removal for the purposes of construction activities are not included in this indicator. This concept is debatable, as sealing land represents a very invasive type of land use in terms of soil conservation, although it is less vulnerable to soil erosion from rainfall.

A comparison of the results obtained with country-specific and global-default CFs are provided in Figure 6.3. The higher impacts for India with the country-specific factor is directly related to higher inventory flows for land use for Indian rice production. For both impact indicators, the differences between the three options are less pronounced when using the global-default CFs. In particular, this can be seen in the case of erosion potential in India, which is more than double in the regionalised case compared with the non-regionalised case. This is due to the fact that the erosion potential CFs are higher for India compared with the global average due to rainfall intensity and topography. This highlights the importance of using country-specific CFs whenever the location of the activities being assessed is known.

6.8. Recommendations and outlook

The recommendations from the task force are broken up into specific recommendations for the choice of characterisation factors, judgement of the quality of these factors and the rationale for the level of recommendation. Further recommendations are then made on good practice for their implementation, inventory requirements, testing the CFs, and future developments.

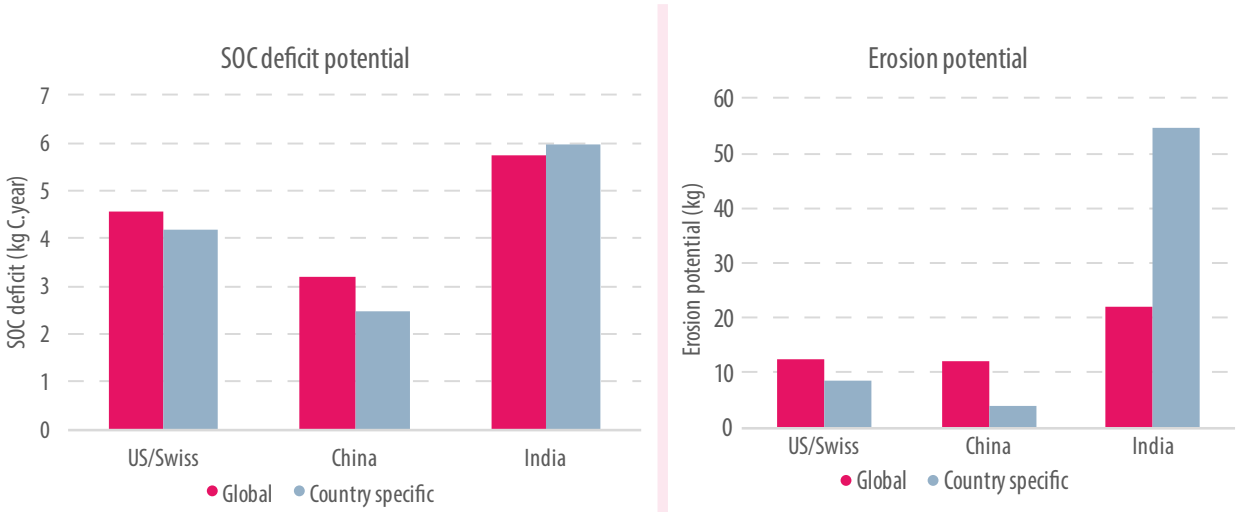

Figure 6.3. Rice case study results for SOC deficit potential and erosion potential using global and country-specific CF.

a) Main recommendations

- An interim recommendation (until necessary CFs are provided, see below) is to use SOC deficit potential based on Brandão & Milà i Canals (2013) with an expanded group of CFs for transformation and occupation of land use types including forests, grasslands, pastures, permanent crops, and artificial or sealed areas[16]. We recommend using CFs for SOC deficit with geographic resolution at the country or subnational (state, provincial, or regional) level where available. We strongly recommend basing CFs for SOC depletion potential on a spatial resolution that is relevant for the LULUC activity under study. For example, national production inventories should use national CFs, while local production may be based on ecoregion level CFs.
- In the absence of a complete model covering all forms of soil loss (wind erosion and water erosion including sheet, rill, gully erosion, and landslides), an interim recommendation is to use erosion potential based on RUSLE as proposed in Bos, Horn et al. (2016) with updates from Horn and Maier (2018), to model water erosion (sheet and rill erosion) after adjusting the reference state to PNV, calculating CFs at the ecoregion level and adding regeneration times for land transformation. These factors will be using regional (most likely national) characterisation factors to the extent permitted in the background LCA databases.
- We suggest that future work link erosion potential to the resource use endpoint accounting for soil dissipation through soil erosion and soil formation relative to the overall global soil resource, or potentially regional soil resources.
- We suggest linking SOC deficit potential indicator to biotic production potential and further to malnutrition and socio-economic assets consistent with approaches being implemented in water footprint links to human health and resources.

b) Judgement on quality, interim versus recommended status of the factors and recommendation

The reasoning for selecting the SOC deficit method is based on the maturity of the model as previously recommended by the ILCD in 2011 (EC JRC 2011) and because it has been used widely since then. SOC is a frequently cited indicator for soil quality that is strongly linked to the soil functions of carbon cycling, nutrient cycling, water retention and pest control, which link to endpoints including biotic production potential and other ecosystem services (Cowie, Orr et al. 2018). However, it is important to note that SOC deficit may be of limited value in areas where SOC is very high or where other important soil threats such as compaction or salinization exist, as increasing SOC in these areas may not enhance endpoints like biotic production and other ecosystem services. The recommendation for SOC deficit is provided as an interim recommendation until CFs are provided for major land use types such as intensive and extensive production forests and perennial crops, which are currently considered as having the same SOC as "natural forests."

Erosion potential is recommended as an additional indicator partly because SOC deficit potential does not have a strong link to the erosion of soil through water erosion. In addition, future work could link erosion potential to its impacts on socio-economic assets including costs for water quality treatment and dredging of reservoirs and rivers. However, in the absence of this link, we have an interim recommendation to use erosion potential as a midpoint indicator.

c) Applicability, maturity, and good practice for factors application

The SOC indicator is relatively mature with five years of use within the ILCD method. The changes proposed in this document are refinements rather than restructuring so the experience gained over this time remains relevant. The land management practices in the IPCC-derived factors (e.g., tillage with low - medium - high input, with and without manure, see Table 6.1) often have no direct correspondence in the elementary flows provided by Koellner, De Baan et al. (2013). It is recommended that users check whether information about tillage practices is available and such CFs may thus better represent the processes in their studied system.

For erosion potential, while the impact method was published in LANCA (2010), the underlying model (revised universal soil loss equation) dates back

16 Minority statement: the choice of SOC as a standalone proxy of soil quality implicitly results in a relevance of zero being given to soil qualities that are not or not well reflected by the SOC and this might not lead to an improved decision support as it risks neglecting relevant aspects.

to 1991 and its predecessor, the universal soil loss equation, dates back to 1965 (Wischmeier and Smith 1965) and is commonly used in soil sciences.

We recommend that both the SOC and erosion CFs are applied in background databases at the national level where the country is known or for states of large nations crossing multiple ecoregions, otherwise at the global level. In the foreground it is suggested that the specific soil type or the ecoregion level where the activity takes place is used where it can be calculated.

Special cases in organic soils (peatlands, organic wetlands): any transformation from these land covers are considered as a shift from the natural state, with the maximum impact. This is already considered in the CFs provided. Also, it should be noted that the ability of SOC to indicate soil quality in organic soils is limited (given that the SOC content is already very high, a slight increase or decrease would not be associated to a significant increase or decrease in quality in organic soils).

It is recommended to use the soil erosion potential in combination with the SOC deficit potential impact category because the indicators consider different impacts. For example, sealed land may reduce soil erosion but cause and increase in SOC deficit potential.

CO_2 emissions related to change in SOC influence climate change; the effects on climate change are not covered by this indicator, but those emissions may significantly affect the GHG emissions caused by the product system at issue. We strongly recommend that they are considered in the climate change impact category. Müller et al. (2012) provide an approach on this aspect.

d) Link to inventory databases (needs for additional inventory features, needs for additional inventory flows, classification or differentiation etc.)

For implementation of the factors in inventory databases most of the land use flows are already available in existing databases. New elementary flow names will be required with country names (and states for large countries) appended to the flow to encode the location until metadata with geolocations is supported in LCA software.

Geographic specificity is a common consideration in other impact categories such as water use and biodiversity impacts from land use activities. Other important flow data relevant for soil includes management practices such as tillage practices for different annual cropping. A correspondence table will be required to link the recommended characterisation factors to existing nomenclature. For example, the characterisation factor for an annual crop will link to all three flows: annual crop, annual crop irrigated, and annual crop non-irrigated.

e) Roadmap for additional tests

The SOC deficit model and erosion models are well understood, however, a number of refinements have been implemented including adjustments to the reference state and inclusion of regeneration times in erosion potential factors from Bos, Horn et al. (2016) and the calculation of country-specific factors for the SOC deficit model. The inclusion of these factors will be checked through a series of existing case studies in relevant production systems, including annual and perennial agriculture and forest products. However, the recommendations are not contingent on completion of these case studies.

f) Next foreseen steps

Adjustment to the characterisation factors needs to be implemented to ensure consistency with the land use elementary flows recommended by the Life Cycle Initiative (Koellner et al., 2013). The correspondence between land use elementary flows in Brandão & Milà i Canals (2013) and Koellner et al. (2013) will be based on the mapping exercise performed by the ILCD (Vidal et al 2017).

In 2019, it is expected that the IPCC will publish revised guidelines that include updated values management factors effecting SOC data, which should be examined for potential inclusion into the current method SOC method used in Brandão & Milà i Canals (2013).

A further development of the CFs provided for both indicators would be to provide them at a smaller geographical scale (e.g., states, ecoregions within a country, based on the coordinates). This would require that LCA software has the capacity to import geo-differentiated CFs, which is discussed in the cross-cutting issues chapter.

Following previous recommendations (Verones et al. 2016) there may also be a need for marginal characterisation factors where the reference land use is based on current land use activities. This option will require some investigation to determine if there is sufficient data to build a global set of characterisation factors, or if it may need to be implemented as part of the foreground of LCA studies.

As mentioned in Section 6.3, an integrative soil quality score could be an interesting option to explore to represent soil quality and its links to relevant endpoints. Kibblewhite, Ritz et al. (2008) highlighted the "*highly integrative pattern of interactions within each of the soil functions*" and proposed a new conception of soil quality based on the maintenance of its key functions. Such a model would be based on assessing directly the results of the soil functions (such as long term biotic production potential, water filtration, etc.) and not the factors involved in the underlying processes, such as SOC (Thoumazeau, Gay et al. 2018). The challenge in this work is to source data on soil quality that could be used to derive a predictive model of integrative soil quality, which is difficult on a global scale.

Calculation of default factors for global crops based on the global distribution of all crops can be undertaken in a similar way to the AWARE water footprint method which has agriculture and non-agriculture CFs (Section 5.5 in Frischknecht and Jolliet [2016]). Where crops grow is influenced by soil and climate conditions so aggregation of factors on a crop rather than geographic basis is both appropriate or feasible from a methodological point of view and practical in that it can be applied to the crop even when the location of the crop is not known. However, the use of CFs in background databases would be limited unless land use definitions include the name of the crop. Otherwise, the CF would be limited for use in the foreground of LCA studies.

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