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Soil quality, properties, and functions in life cycle assessment: an evaluation of models

B. Vidal Legaz [a], D. Maia De Souza [b, c, d], R.F.M. Teixeira [e], A. Antón [f], B. Putman [g], S. Sala [a, *]

[a] European Commission, Joint Research Centre, Institute for Environment and Sustainability, Sustainability Assessment Unit, Via Enrico Fermi 2749, 21027 Ispra (VA), Italy
[b] Swedish University of Agricultural Sciences, Department of Energy and Technology, Uppsala, Sweden
[c] University of Alberta, Department of Agricultural, Food and Nutritional Science, Edmonton, Canada
[d] Agriculture and Agri-Food Canada, Lethbridge Research Centre, Lethbridge, Canada
[e] MARETEC — Marine, Environment and Technology Centre, Instituto Superior Técnico, Universidade de Lisboa, Av. Rovisco Pais, 1, 1049-001 Lisboa, Portugal
[f] IRTA, Barcelona, Spain
[g] University of Arkansas, Department Biological and Agricultural Engineering, Fayetteville, AR, USA

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A B S T R A C T

Soils provide essential ecosystem services for supporting both human and ecosystem needs and has been under pressures resulting from the intensification and expansion of human activities. In the last 15 years, substantial efforts have been made to quantify the impacts on soils derived from production systems and their related supply chains. In this study, a systematic, qualitative evaluation of up-to-date models connecting land occupation and land transformation to soil impact indicators (e.g., soil properties, functions, and threats) is performed. The focus is on models that may be applied for assessing supply chains, namely in the context of life cycle assessment (LCA). A range of eleven soil-related models was selected and evaluated against different criteria, including scientific soundness, stakeholders' acceptance, reproducibility, and the applicability of models from the perspective of LCA practitioners. Additionally, this study proposes a new land use cause-effect chain to qualify the impacts of land use on soils. None of the models is fulfilling all the criteria and includes comprehensively the cause-effect impact pathways. Notably, trade-offs were most frequent between the relevance of the modeled impact processes and the models' applicability. On the one hand, models proposing multi-indicators cover several drivers of impacts and have a broader scope. On the other hand, several models just focus on one driver of impact, but may provide more relevant impact characterization. Our results provide common ground for the development and identification of models that provide a comprehensive and robust assessment of land use change and land use impacts on soils. Indeed, to ensure both a comprehensive and relevant characterization of impacts, the study identifies several research needs for further models' developments, namely: 1) adopting a common land use cause-effect chain and land use classification; 2) accounting for different land management and land use intensities; 3) expanding the inventory data beyond the accounting of the area related to a certain land use; 4) assessing the added value of multi-indicators compared to single indicators, including the reduction of possible redundancies in the impact evaluation; 5) improving consistency from midpoint to endpoint characterization, especially the link with biodiversity; 6) guiding the calculation of normalization factors; and 7) assessing systematically model's uncertainty.

1. Introduction

Soil quality is defined as the "capacity of a living soil to: function, within natural or managed ecosystem boundaries; to sustain plant and animal productivity; to maintain or enhance water and air quality; and to promote plant and animal health" (Doran, 2002). This concept is closely related to soils capacity to deliver essential ecosystem services such as freshwater purification and regulation (Garrigues et al., 2012), food and fiber production, and the

* Corresponding author.
E-mail address: serenella.sala@jrc.ec.europa.eu (S. Sala).

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maintenance of global ecosystem functions. Ensuring the maintenance of high quality standards for the state of soils is, therefore, a fundamental requirement for global sustainability (Doran, 2002). The intensification and expansion of human activities have placed increasing pressure on land resources, resulting in soil quality degradation, particularly linked to land use and land use change (MEA, 2005). A report on the status of world's soil resources (FAO, 2015) shows the majority of soils are in fair, poor, or very poor conditions. Some of the most worrisome conditions are characterized by advanced degrees of erosion, leading to crop losses, and increased soil acidity with a lack of soil nutrients, constraining food production (Blum, 2005; Menon et al., 2014). For instance, the condition of soils in the Middle East and North Africa is generally considered to be very poor as a result of advanced soil erosion, sealing, and contamination; while in Europe, soils are considered poor as a result of poor nutrient balance, acidification, soil sealing, contamination, and salinization. The trend towards soil degradation is expected to continue, with projected increases in livestock production (Bouwman et al., 2013), deforestation rates, poor water and nutrient management, and large-scale applications of pesticides (FAO and ITPS, 2015). The FAO report also identifies the need to understand the spatial and temporal variations in soil functions as well as monitor soil changes. Accordingly, attention has been given to the development of indicators for monitoring pressure on soil due to human activities (Van Oudenhoven et al., 2012; Niemi et al., 2015). Consequently, there is a clear need to assess the extent to which soil quality is affected by current human interventions (Jandl et al., 2014), and to detect hotspots along supply chains as well as possible “sustainable land management” options (Liedtke et al., 2010; Del Borghi et al., 2014). Even so, quantifying impacts on soil functions is challenging given the complexity of soil processes and the spatial and temporal variability of soil properties (Lauber et al., 2013; Vereecken et al., 2014). Accounting for this variability determines the adequacy of a soil quality model to represent local conditions (Doran and Parkin, 1996).

In the literature, three main quantitative approaches to the land footprinting, i.e. the impact assessment of human pressures on land, were identified: 1) mere land accounting, which reports the area of land use associated with certain activities/crops (e.g. expressed in m^2); 2) weighted accounting, which estimates the amount of land standardized to factors as the productivity of the land, e.g., the ecological footprint (Wackernagel, 2014); and 3) the quantification of the change of a specific soil feature resulting from land interventions, e.g., soil organic matter (Milà i Canals et al., 2007a, 2007b).

In light of these approaches, an integrated assessment method is needed to assess and allocate the impacts of specific production systems (at the product level) on natural resources like soil. Life Cycle Assessment (LCA) (ISO, 2006a, 2006b) is considered one of the best approaches to quantify the potential impacts of production from a life cycle perspective (Hellweg and Milà i Canals, 2014). In LCA, potential impacts can be assessed by two types of indicators. On the one hand, endpoint or damage indicators address aspects to safeguard, denominated Areas of Protection (AoP) in an LCA context, i.e., the natural environment (e.g. biodiversity), natural resources (e.g. resource availability), and human health (e.g. life expectancy of humans). On the other hand, LCA might also include midpoint indicators, which are intermediate aspects between the life cycle inventory (LCI) —e.g., the amount of pollutants emitted, resources used, or land use associated with production processes— and the endpoints.

In the last 15 years, in the LCA community, substantial efforts have been made to improve the assessment of impacts due to land use. However, due to the challenges of quantifying these impacts (Li et al., 2007), soil properties and functions have been incorporated in a very limited way. Midpoint indicators have usually consisted of the sum of the area of land occupied and/or transformed for the production of a certain amount of product. This type of data is generally available in LCA software and inventories. However, data on only the amount of land used is an inappropriate basis for comparisons of products (Helin et al., 2014), and the assessment of land use impacts needs to be more inclusive (Koellner et al., 2013a). Indeed, according to the United Nations Environmental Programme—Society of Environmental Toxicology and Chemistry (UNEP-SETAC) Life Cycle Initiative LCA, land use models should focus both on soil quality, biotic production, and biodiversity. Several endpoint indicators have generally focused on the damage to biodiversity caused by land use (e.g. loss in species' richness as in De Baan et al., 2013; Souza et al., 2015). However, a consensus on the best available model for impact on biodiversity due to land use is difficult to be achieved (Teixeira et al., 2016), as demonstrated in a parallel review conducted by the UNEP-SETAC Life Cycle Initiative task force on land use impacts on biodiversity (Curran et al., 2016).

Regarding midpoint indicators, so far efforts to select a model to be widely applied has been made mainly by the European Commission, which in 2011 assessed several models and recommended the model developed by Milà i Canals et al. (2007a; 2007b) within the International Reference Life Cycle Data System handbook (ILCD handbook) (EC-JRC, 2010; 2011). This has been further adopted also as model for the product environment footprint (PEF) (EC, 2013). This model was selected as a result of an evaluation of land use models (developed up to 2009) against defined criteria (inspired by those of ISO, 2006b and related to: environmental relevance, applicability, robustness, etc). Specifically, the recommended model adopts soil organic matter (SOM) as a stand-alone indicator for the assessment of land use impacts at the midpoint level. Notably, important soil functions are disregarded in this model, even though SOM is considered one of the most important indicators for the sustainability of cropping systems (Fageria, 2012) and plays a crucial role in provisioning (e.g. biotic production) and supporting services (e.g. climate regulation). Examples of these ignored functions include resistance to erosion, compaction, and salinization (Mattila et al., 2011). Therefore, the model was considered not fully satisfactory and was recommended to be applied with caution (EC-JRC, 2011).

Internationally, scientific efforts of the UNEP-SETAC Life Cycle Initiative have resulted in a harmonized classification of land use/cover types (Koellner et al., 2013b) to guarantee a better coverage of land use typologies and improve the comparability of modeling results.

In spite of these efforts, several issues are still critical and may affect modeling assumptions and results (Allacker et al., 2014). First, a clear and consistent cause-effect chain, also called *impact pathway*, is still missing. The impact pathway should depict systematically the causal relationships from the inventory data (amount and typology of land use) to the mid- and endpoint indicators and areas of protection. Second, current models that could be applicable in LCA are unable to comprehensively depict the multiple impacts derived from land use and land use change. Moreover, many of these models are originally based on site-specific studies (and data) and require additional effort for their adaptation to other locations and spatial scales. Finally, the reference state used to assess the potential environmental impacts often differs among models, making results incomparable.

As a result of this lack of inclusiveness in the nuances of soil quality in models, there is a need to improve the available models, ensuring their wider applicability in LCA and their comprehensiveness in modeling the key drivers of impacts on soil quality. To fill this gap in research, this paper reviews the models that assess potential land use impacts on soils at midpoint level. Specifically, this study highlights the main characteristics of the available impact characterization models, and it identifies future research needs. The analysis builds and expands the review done by EC-JRC, 2011. Hence, the ILCD criteria are adopted and the ILCD nomenclature is taken as reference for the applicability.

The paper is organized as follows: first the methods for assessing the different available models are presented, including a brief overview of the evolution of the assessment of soil quality and functions in LCA (section 2); next, the results of the review and assessment of models are presented (section 3) as basis for discussing the status quo and the research needs in the field (section 4).

2. Methods for assessing land use models in LCA

The assessment of the different land use models available in LCA included several steps:

- overview of state of the art on the modeling of soil quality and functions in LCA (section 2.1);
- review of the literature and pre-selection of models for evaluation (section 2.1.1);
- definition of an impact pathway that reflects the latest knowledge in the cause-effect chain for land interventions (section 2.2);
- definition of a set of criteria and sub-criteria for the evaluation of models (section 2.3);
- evaluation of the models (presented in the results, section 3).

2.1. Soil quality and soil functions in LCA

The first efforts to address impacts on soil properties and functions in LCA date back to the 1990's when Heijungs et al. (1997) proposed an evaluation of land use based on the occupation of land by a production system. This first attempt disregarded the state of the soil, its supporting services, and the properties that frequently depend non-linearly on the spatial and temporal scale of the intervention. Following this, Cowell and Clift (1997, 2000) proposed the first model that assessed soil quality and quantity in LCA, as a way to address the impacts of agricultural and industrial systems. They proposed five groups of factors, divided into three midpoint impact categories (soil erosion, compaction, and change in organic matter). These groups of factors were 1) mass of soil, 2) living organisms (weeds/weed seeds and pathogens), 3) trace substances (nutrients, salts, and pH of soil), 4) non-living matter (organic matter), and 5) form of soil (texture and structure).

A different set of indicators was proposed by Mattsson et al., 2000, aiming at evaluating the sustainability of the production capacity of soil in three different types of vegetable oil crops (rape seed, palm oil, and soybean), in three distinct regions (Sweden, Malaysia, and Brazil, respectively). This model considered soil erosion, hydrology effects, and other indicators for the assessment of soil quality, e.g., SOM, soil structure, and pH. Additionally, the nature value of soil was addressed by Weidema and Lindeijer (2001), who proposed five indicators: 1) substance and energy cycles; 2) productivity; 3) biodiversity; 4) cultural value; and 5) migration and dispersal. In their proposed impact pathway, they included “altered soil functions” as a midpoint category, linking it to life-support functions.

Over the years, within the context of selecting soil indicators, other models addressing land use impacts through multiple indicators have been proposed (LANCA - Beck et al., 2010; Saad et al., 2013; SALCA-SQ - Oberholzer et al., 2012), in comparison to e.g. the model of Milà i Canals et al. (2007a; 2007b), which proposes the use of a single indicator. Additionally, models following a more qualitative approach have been developed proposing a holistic approach in characterizing soil quality. Examples of the latter are the multi-indicator model SALCA-SQ (Oberholzer et al., 2012) based on a qualitative scoring system, very detailed and comprehensive regarding the list of soil aspects accounted for. With a different focus, another qualitative approach is the hemeroby index (Brentrup et al., 2002). The hemeroby is a measure of naturalness, i.e. calculates the magnitude of deviation from the potential natural vegetation (these naturalness categories have been recently refined by Fehrenbach et al., 2015).

It is increasingly evident that the selection of a specific indicator (or a set of indicators) is problematic, given the spatial and temporal variability of soil quality (Milà i Canals et al., 2007a; Garrigues et al., 2012). Additional challenges are associated with the level of detail required in LCA inventory data to reach accurate results with an acceptable spatial resolution. Also, the state of the soil as well as the threats to soil should be considered in the selection of proper indicators. For example, the European Thematic Strategy for Soil Protection (EC, 2006) lists soil erosion, organic carbon decline, compaction, salinization, contamination, acidification, and landslides as the main soil threats. Indeed, focusing on LCA studies, Garrigues et al. (2012) state that models should incorporate what they consider as the main threats to soil and to its functions, e.g., erosion, decline in SOM, biodiversity loss, contamination, sealing, compaction, and salinization. They also state that midpoint indicators on soil-related threats can be more easily related to the production units in an LCA context than indicators on soil properties.

2.1.1. Selection of models

In this study, land-use models previously evaluated in the ILCD handbook (EC-JRC, 2011) were examined for modeling of land use impact at the midpoint level, with a focus on soil-related indicators. Subsequently, a review of the literature was conducted to incorporate models that assess soil properties/functions/threats not previously considered in the ILCD handbook, i.e., models developed after those assessed in the ILCD up to year 2009. In parallel, model developers were contacted when a more profound understanding of their models was required.

This study identified 31 models, which were shortlisted for further evaluation only if they: 1) computed indicators for assessing soil properties/functions/threats; 2) were compatible with LCA, i.e., they were used to calculate impact indicators starting from elementary flows presented in LCI; 3) produced an output consisting of characterization factors (CFs), i.e., the parameters that allow deriving impact (midpoint/endpoint) indicators from the LCA inventory data, or an output that could be easily converted into CFs. A total of 11 models fulfilled these requirements and further underwent the evaluation procedure (Table 1), whose description follows here.

The first model evaluated was an updated version of the model currently recommended in the ILCD handbook (Milà i Canals et al., 2007a, 2007b), which includes Soil Organic Carbon (SOC) as a stand-alone soil quality indicator (Brandão and Milà i Canals, 2013). SOC is used as a way to approach the productive capacity of the soil, which in turn may affect two areas of protection: natural resources and natural environment. Unlike the previous version of the land use framework (Milà i Canals et al., 2007a, 2007b), which characterized impacts only in the United Kingdom, Brandão and Milà i Canals (2013) provided CFs for a global application of the model. Subsequently, three multi-indicators models were included that evaluated soil properties and functions: LANCA (Beck et al., 2010; Bos et al., 2016); Saad et al. (2013); and SALCA-SQ (Oberholzer et al., 2012).

Table 1
Overview of models evaluated in this study.

Model	Main indicators	Unit (for land occupation impact)
Brandão and Milà i Canals (2013)	- Soil Organic Carbon (SOC) – as indicator of Biotic Production Potential (BPP)	tC-year/(ha-year)
LANCA- Land Use Indicator Value Calculation in Life Cycle Assessment (**Beck et al., 2010**; CFs associated to land use flows developed by **Bos et al., 2016**)	- Erosion resistance - Mechanical filtration - Physicochemical filtration - Groundwater replenishment - Biotic production	kg/m^2 year m^3/m^2 year mol/m^2 m^3/m^2 year kg/m^2 year
Saad et al. (2013)	- Erosion resistance - Mechanical filtration - Physicochemical filtration - Groundwater recharge	t/(ha year) cm/day $cmol_c/kg_{soil}$ mm/year
SALCA-SQ **Oberholzer et al. (2012)**	- Soil properties indicators: rooting depth, macro-pore volume, aggregate stability, organic carbon, heavy metals - organic pollutants, earthworm biomass, microbial biomass, microbial activity - Impact indicators: risk of soil erosion, risk of soil compaction	(many different)
Núñez et al. (2010)	- Desertification index	dimensionless
Garrigues et al. (2013)	- Total soil area compacted - Loss of pore volume	m^2/ha, m^2/t m^3/ha, m^3/t
Núñez et al. (2013)	- Emergy - Net Primary Production (NPP) depletion	MJse g^{-1} soil loss m^2 year
Alvarenga et al. (2013)	- Exergy of natural land (biomass extraction-based) - Exergy of human-made land (potential NPP-based)	MJ ex/m^2 year
Alvarenga et al. (2015)	- Human Appropriation of NPP (HANPP)	kg dry matter/m^2 year
Gardi et al. (2013)	- Soil pressure (on biodiversity)	
Burkhard et al. (2012)	- Ecosystem integrity indicators (7) - Ecosystem services indicators (22) - Demand of ecosystem services (22)	dimensionless (ranking)

LANCA is an updated version of the model developed by Baitz (2002), which was already evaluated in the ILCD handbook. LANCA calculates indicators for soil functions, e.g., erosion and water regulation and filtration capacity, originally based on site-specific data. Although not recommended by ILCD, partly because of the lack of CFs and the large amount of input data required, LANCA has been added here as their developers have recently calculated CFs directly associated with land use flows (Bos et al., 2016). They developed a global application of the LANCA model including some minor methodological modifications and deriving the indicators directly and solely from land use inventory flows. Saad et al. (2013) developed their model based on LANCA, selecting four impact indicators to address ecosystem services damage potential. Finally, the SALCA-SQ model, based on site-specific data, focuses on soil properties, e.g., macropore volume and microbial activity, and threats to soil, e.g., erosion and compaction. Notably, LANCA and SALCA-SQ do not establish explicit links to endpoint indicators or AoPs.

Next, three threat-specific models were included (Núñez et al., 2010; Garrigues et al., 2013; and Núñez et al., 2013). Núñez et al. (2010) calculated a desertification index based on aridity, erosion, aquifer over-exploitation, and fire risk. Garrigues et al. (2013) focused on soil compaction as a result of the use of agricultural machinery. They calculated auxiliary indicators such as water erosion and SOM change. The model of Garrigues et al. (2013) is meant to be part of a broader framework, which should include other processes, e.g., erosion, change in SOM, and salinization. Both of these models use indicators that assess the capacity of the soil to provide ecosystem services and support biodiversity, although these linkages are not explicitly addressed by the authors. Núñez et al. (2013) compute the loss of Net Primary Production (NPP) and emergy as indicators of damage to the AoP natural environment (ecosystems) and resources, respectively. Both indicators are based on soil loss calculated through the application of the USLE erosion equation (Wischmeier and Smith, 1978); while the NPP depletion is calculated as a function of SOC loss the emergy indicator, which expresses all the energy embodied in the system, is derived from soil loss. All three models show limitations regarding the availability of CFs in that: CFs for Núñez et al. (2013, 2010) are not related to land use inventory flows, and CFs for Garrigues et al. (2013) are site-specific.

Furthermore, two models based on thermodynamics accounting were pre-selected: exergy-based accounting of soil as resource (Alvarenga et al., 2013), and the Human Appropriation of Primary Production (HANPP) (Alvarenga et al., 2015). Alvarenga et al. (2013) computed exergy distinctly for natural and human-made land. Exergy of extracted biomass was calculated for natural land covers, while the exergy associated to potential NPP was used for human-made land. Alvarenga et al. (2015) focused on the amount of NPP that is not available for nature due to human use of land (HANPP). Both exergy and HANPP, as stated by the authors, model impacts on the AoP natural resources and natural environment. The CFs resulting from both models are directly associated with land use flows.

Finally, two models not specifically developed for LCA were selected given their potential applicability in the LCA context (Gardi et al., 2013; Burkhard et al., 2012). Gardi et al. (2013) developed a composite indicator of pressures on soil biodiversity, which is a weighted index of variables related to land use (agriculture intensity, land use change), threats to soil (compaction, erosion, contamination, SOC loss), and threats to biodiversity (invasive species). This index may serve to reveal impacts on the AoP natural environment. Comparatively, Burkhard et al. (2012) provide a model to score land use types according to a set of ecosystem integrity and ecosystem services indicators. These indicators include, among others, soil functions, water provision, biodiversity loss, and exergy. The scores are based on expert judgment and case studies. Notably, the model does not establish any link between indicators, but rather calculates them directly and solely associated with each land use type. This model also includes endpoint indicators among ecosystem services indicators.

2.2. Impact pathway for land use

The LCA land use impact pathway for land use was revisited (Fig. 1), aiming at identifying indicators that are not yet covered and unravel unclear links between the LCI data, midpoint and endpoint indicators. This new impact pathway was built considering the last developments regarding land use impact assessment in LCA (Milà i Canals et al., 2007c; Koellner et al., 2013b; Souza et al., 2015; Curran et al., 2016); and impact pathways of pre-selected models (Garrigues et al., 2013; Núñez et al., 2013). Among those studies, Curran et al. (2016) proposed an impact pathway with a focus on biodiversity, partially considering indicators of impacts on soil quality and habitats.

The impact pathway proposed here (Fig. 1) starts from the different properties and functions of the soil related to the geomorphological and pedological features of soils before any land intervention. Soil functions generally refer, among others, to the soil's capacity to supply nutrients to plants (soil fertility), and to regulate water flow and erosion. Functions such as the provision of habitat also depend on the land spatial structure, i.e., land configuration, including the natural/human-made vegetation mosaic that characterize a landscape, e.g. the presence of hedgerows. These initial soil conditions, associated with the nature and intensity of land interventions, will determine the impacts on soil. The latter can be measured by different indicators of soil degradation, namely: erosion, sealing, SOC change, compaction, or contamination. The spatial structure of the land might also be affected by the fragmentation of the landscape. All these threats to soil have an impact on the soil capacity to supply ecosystem services and may affect the three AoPs. The impacts of land use might also be estimated by changes in ecosystem thermodynamics, which may be accounted for by e.g., exergy or emergy accounting.

2.3. Criteria for the evaluation of models

A set of criteria was developed to evaluate the models building on those defined in the ILCD handbook (EC-JRC, 2010) and in Curran et al. (2016). These criteria build on the rationale of the selection of models and indicators proposed in ISO (2006b), expanding and detailing them specifically for land use. The set of criteria includes a descriptive section (summary information), followed by an evaluation section in which models were qualitatively assessed against: 1) completeness of the scope, 2) environmental relevance, 3) scientific robustness and uncertainty, 4) documentation,

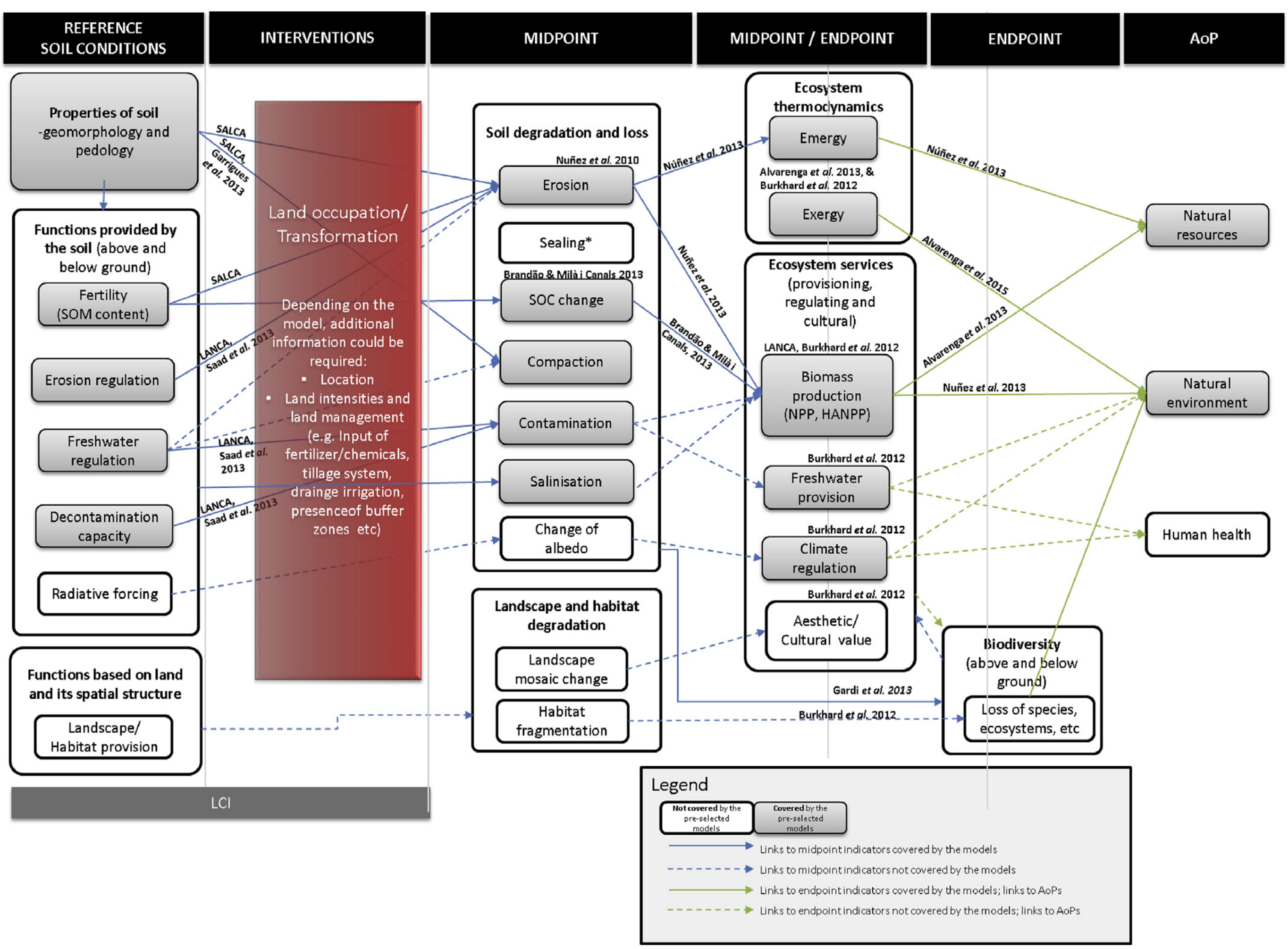

Fig. 1. Reviewed impact pathway for land use. The positioning of the pre-selected models is displayed as: Brandão and Milà i Canals, 2013; LANCA (Beck et al., 2010; Bos et al., 2016); Saad et al., 2013; SALCA (Oberholzer et al., 2012); Núñez et al., 2010; Garrigues et al., 2013; Núñez et al., 2013; Alvarenga et al., 2013; Alvarenga et al., 2015; Gardi et al., 2013; and Burkhard et al., 2012. The soil threat salinization as such is missing, which is considered under "contamination". Some indicators could be interpreted either as midpoint or endpoint, depending on what AoP they are associated with – e.g. biomass production would be an endpoint indicator when focusing on the AoP natural resources, but a midpoint indicator when referring to the AoP natural environment, on which endpoints (e.g. biodiversity) will rely on.

transparency, and reproducibility, 5) applicability, and 6) stakeholders' acceptance. Scores between A and E were assigned, which generally mean, respectively, the best (A) and the worst (E) possible results. The criteria are listed in Table 2, while the Supplementary Material (SM 1) includes the complete evaluation form and the scoring rules. A brief description of the evaluation criteria is provided next.

Summary information. It consists of relevant background information: the purpose, theoretical foundation of the model and the indicators adopted; the impact pathway; the modeling reference state and time frame; and the land use inventory flows used by the model.

Completeness of the scope. It was assessed whether the proposed midpoint indicator(s) cover(s) relevant information related to the modeled impacts, including how close the indicators are to inventory data. Lastly, the AoPs and geographic coverage, e.g., local, regional or global, of each model were evaluated.

Environmental relevance. These set of criteria evaluates the comprehensiveness of models and their capacity to discriminate the impact of different types of land use. The comprehensiveness aspect encompasses the coverage of land use inventory flows as proposed by ILCD, which follows from the land use classification proposed by Koellner et al. (2013b). This ILCD land use classification includes a complete coverage of land use types, and aggregates them in four hierarchical levels. This criterion also assesses if the model addresses land use's specific aspects: 1) the distinction between extensive and intensive land uses; 2) if the model considers the impacts of both land occupation and transformation; and 3) the type of soil indicators included, e.g., soil properties, functions, and threats. Finally, the spatial-temporal resolution of the model input

Table 2
Criteria and sub-criteria for the evaluation of the models. See Supplementary Material SM 2 for further details.

Description		Aspect
Summary information (descriptive)		• Purpose and interpretation • Description of the impact pathway • Impact categories covered • Midpoint indicators • Reference state and time frame • Underlying model(s) • LCI flows covered
Evaluation criteria		**Sub-criteria**
Completeness of scope		• Impact pathway consistency between mid- and endpoint • Midpoint indicator placement • AoP coverage • Geographic coverage
Environmental relevance	Comprehensiveness	• Coverage of ILCD LCI flows • Distinction between extensive and intensive use • Land occupation/transformation impact • Permanent impact • Direct/indirect land use change • Soil quality • Soil threats • Land availability • Biotic production • Erosion regulation • Water regulation functions • Biodiversity support • Climate change • Aesthetic/cultural value
	Spatial-temporal resolution	• Temporal resolution of the model input • Spatial resolution of the model input
Scientific robustness and uncertainty	Scientific robustness	• Characterization model peer review • Value choices robustness • Up-to-date knowledge for the cause-effect chain
	Uncertainty	• Quality of the model input data • Uncertainty assessment
Documentation, transparency and reproducibility		• Accessibility of the documentation • Completeness of the documentation • Accessibility of the input data • Accessibility of the characterization factors • Accessibility of the model • Modeling assumptions and value choices
Applicability		• Compatibility with LCA methodology • Availability of LCI datasets • Usability of characterization factors • Availability of normalization factors • Ease to update to conform ILCD • Temporal resolution of characterization factors • Spatial resolution of characterization factors • Spatial resolution of LCI flows
Stakeholders acceptance		• Model and model results • Authoritative body • Academic authority • Neutrality across industries, products or processes. • Relevance for current policy

data was evaluated as a determinant aspect of the accuracy of the model results.

Scientific robustness and uncertainty. The criteria of scientific robustness and uncertainty specifies if the model has been peer-reviewed; if there are already case studies that can validate the soundness of the model; the robustness of model choices; whether it is based on up-to-date knowledge and quality data; and if uncertainty was reported.

Documentation, transparency and reproducibility. It considers the accessibility and the completeness of model documentation, CFs and input data.

Applicability. As models may be scientifically relevant and comprehensive but very difficult to be implemented in the practice of an LCA study, the compatibility of the model with the LCA method was assessed. This includes: the availability of the required inventory flows; the level of implementation in LCA software; the availability of CFs and normalization factors; and the ease to conform to ILCD. ILCD has been selected as representative nomenclature for ensuring that the set of computed indicators are able to be used by an LCA practitioner. The spatial-temporal resolution of the CFs and inventory flows was also assessed. Whenever available, CFs of the reviewed models were compiled, and they are provided in the Supplementary Material (SM 3), adapted by this study to the ILCD nomenclature.

Stakeholders' acceptance. The understandability of the model results and its uncertainty were evaluated, as well as the model endorsement by academic/authoritative bodies. The models' neutrality and relevance to policy were also assessed.

3. Results

In the following sub-sections the results are reported by model (section 3.1) and by criteria (section 3.2). Table 3 summarizes the scores obtained in the evaluation by each model, while the complete evaluation results are provided in the Supplementary Material (SM 2).

3.1. Evaluation results by model

The main insights of the evaluation of each model are provided in the following.

Brandão and Milà i Canals (2013). This model, which calculates the impacts on the AoP *natural resources*, scored well in terms of impact characterization and applicability. It provided CFs both for land occupation and transformation on a global scale and by climatic region, focusing mainly on impacts related to agricultural production, including the differentiation between extensive and intensive land uses. The scientific robustness of this model was adequate, although the uncertainty assessment and validation of the estimates with field data were missing. Documentation of the model was complete and transparent. As for applicability, land use flows differ from the ILCD nomenclature.

LANCA and Saad* et al. *(2013). Being Saad et al. (2013) based on LANCA (Beck et al., 2010), both models showed particularly good results for the criteria in assessing environmental relevance and applicability. LANCA and Saad et al. (2013) pertain to the multi-indicator model typology, with LANCA potentially covering both the AoPs natural environment and natural resources, while Saad et al. (2013) covered only the former due to the absence of the biotic production indicator. Both models provided CFs, and obtained good results in terms of environmental relevance. Aspects contributing to that were the capacity of the models to compute the impact of both occupation and transformation, and to compute the indicators directly associated with land use inventory data. With regard to these inventory data, LANCA in its latest version of characterization factors (Bos et al., 2016) covered a much larger set of land use types than Saad et al. (2013), which covers flows only at a very high hierarchical level, e.g., shrubland, forest. The underlying sets of models used on both models were considered robust including how value choices were addressed. However, both models showed room for improvement in terms of being up-to-date since some of the underlying models were developed in the 1980's and their revisions are not cited in the documentation. Case studies applying the LANCA model have been undertaken. However, with the latest set of characterization factors (Bos et al., 2016) none has been developed yet. The study by Saad et al. (2013) presented a theoretical approach, which stated that the model was limited due to the coarse scale of application. As for the assessment of uncertainty, while statistical figures are not available for LANCA, Saad et al. (2013) analyzed the variability of resulting CFs when changing the spatial scale used for their regionalization. Complementarily, the influence of input data on the model results was assessed in a previous study (Saad et al., 2011), where an application for Canada was conducted. Like most of the evaluated models, LANCA and Saad et al. (2013) count on good and accessible documentation. Conversely, the lack of access to the model itself, due to its private ownership, was a limitation, in conjunction with difficulties in accessing the input data required for the site-specific application of LANCA.

SALCA-SQ. This model assesses the impacts of agricultural land use on the soil in a qualitative manner (using the following scoring: –, -, 0, +, ++). It covers a very detailed set of soil properties and indicators. Its main limitation regarding completeness of scope was its foundation in site-specific data, and its focus on solely agricultural land uses within a European context. The model's capacity to distinguish different impacts on soil properties is high, based on extensive input data related to specific land management practices. However, the power to provide CFs with a relevant differentiation of the different level of the impacts is limited since CFs are provided only on a qualitative scale. Moreover, no guidance was provided on how to assess the importance of each of the multiple indicators. The scientific robustness of this model was judged to be acceptable as it was peer reviewed and scientifically validated by a case study. However, no estimates of uncertainty for the midpoint indicators were included, and the authors mentioned that not all observed impacts were consistent with model results. Notably, the documentation of the characterization model was accessible and transparent. Regarding the applicability, farm-level data was required, which is often difficult to come by and would require considerable effort to implement.

Núñez* et al. *(2010). This model proposes a desertification index linked to three AoPs (natural environment, natural resources, and human health). It provides CFs for all ecoregions in the world, but without distinguishing among different land use types or land management practices. Therefore, the CFs cannot be applied to derive the impact of the different land use types. The documentation was transparent and all the information required for the desertification impact assessment was generally available.

Garrigues* et al. *(2013). This model presents significant limitations on its applicability in LCA, namely: missing readily calculated CFs; requiring very detailed input information on the production system management (e.g. type of crop and data on machine operations), soil type and local/regional climate. At the moment, only a set of CFs covering some crops in three countries (France, Brazil and Pakistan) is available. Thus, although modeling assumptions and value choices are well documented, the same does not apply for the CFs. Nevertheless, the requirement of local and regional data attributes the model a higher spatial resolution, i.e., a higher accuracy in the prediction of potential impacts.

Núñez* et al. *(2013). This model, which estimates the erosion

Table 3
Results of the models' evaluation. The scores of each criterion assessed within each of the five evaluation criteria groups is reported, ranging between A and E − which generally mean, respectively, the best and worst possible results. The complete evaluations, which include also additional details explaining the scoring assigned, are provided in the Supplementary Material (SM 2).

	Criteria	Brandão and Milà i Canals 2013	LANCA (Beck et al. 2010; Bos et al. 2016)	Saad et al. 2013	SALCA (Oberholzer et al. 2012)	Núñez et al. 2010	Garrigues et al. 2013	Núñez et al. 2013	Alvarenga et al. 2013	Alvarenga et al. 2015	Gardi et al. 2013	Burkhard et al. 2012
Completeness of the scope	• Impact pathway consistency between mid- and endpoint	C	C	C	C	E	E	A	A	A	A	C
	• Midpoint indicator placement in the impact pathway regarding LCI flows	A	A	A	A	A	C	A	A	A	A	A
	• AoP coverage by the midpoint characterization model	C	A	C	C	C	C	C	C	C	C	A
	• Geographic coverage	A	A	A	E	A	E	A	A	A	D	E
Environmental relevance	• Relevance of the characterization	A	C	C	C	C	C	C	E	C	A	C
	• Coverage of ILCD LCI flows	C	A	E	E	E	E	E	E	C	E	C
	• Distinction extensive/intensive use	A	A	E	C	E	C	E	C	C	A	C
	• Land occupation/transformation impact	A	B	A	D	D	D	D	D	C	D	D
	• Permanent impact	A	A	A	E	E	E	E	E	E	E	E
	• Direct/indirect land use change	E	E	E	E	E	E	B	E	E	E	E
	• Soil quality	A	A	A	A	E	C	C	E	E	A	A
	• Soil threats	C	A	A	A	C	C	C	E	E	A	A
	• Land availability	E	E	E	E	E	E	E	E	E	E	A
	• Biotic production	C	A	E	E	E	E	C	A	A	E	A
	• Erosion regulation	E	A	A	A	C	E	A	E	E	A	A
	• Water regulation functions	E	A	A	E	C	A	E	E	E	E	A
	• Biodiversity support	E	E	E	C	E	E	E	E	C	A	A
	• Climate change	E	E	E	E	E	C	E	E	E	E	E
	• Aesthetic/cultural value	E	E	E	E	E	E	E	E	E	E	A
	• Temporal resolution of the input	D	A	D	D	D	D	D	D	D	D	D
	• Spatial resolution of the input	D	A	B	A	D	A	A	A	C	A	B
Scientific robustness and uncertainty	• Characterization model peer review and (peers) acceptance	B	B	A	B	B	B	B	B	B	A	B
	• Value choices robustness	C	B	B	B	B	B	B	B	B	A	B
	• The model reflects the up-to-date knowledge for the cause-effect chain	C	C	C	C	C	A	C	C	A	A	C
	• Presence of an application of the CFs in a case study	B	B	D	B	B	B	B	B	B	E	E
	• Quality of the model input data	C	C	C	C	E	C	C	E	E	C	C
	• Uncertainty assessment	D	E	C	E	E	E	D	C	C	A	E
Documentation, transparency and reproducibility	• Accessibility of the characterization model documentation	A	A	A	A	A	C	A	A	A	C	A
	• Completeness of the characterization model documentation	A	A	A	A	A	A	A	A	A	C	C
	• Accessibility of the input data	A	C	C	C	B	C	A	A	A	C	C
	• Accessibility of the characterization factors	A	A	A	E	A	E	A	A	A	E	C
	• Accessibility of the characterization model	A	E	E	C	E	D	E	C	C	E	E
	• Modeling assumptions and value choices	A	C	C	C	A	A	A	A	A	A	C
Applicability	• Compatibility with LCA methodology	A	A	A	A	A	A	A	A	A	C	C
	• Availability of LCI datasets for the application of the model	C	A	A	C	C	C	C	A	A	C	A
	• Usability of characterization factors for LCA practitioners	C	A	A	E	E	E	E	C	C	E	C
	• Availability of normalization factors for LCA practitioners	E	E	E	E	E	E	E	E	E	E	E
	• Ease to update to conform with the ILCD nomenclature and units	C	C	C	E	D	E	D	C	C	E	C
	• Temporal resolution of characterization factors	D	D	D	E	D	D	D	D	D	D	A
	• Spatial resolution of characterization factors	E	A	E	A	E	C	C	C	C	A	E
	• Spatial resolution of LCI flows	E	A	E	A	E	A	C	A	C	A	A
Stakeholders' acceptance	• Model and model results	A	E	E	C	C	C	C	C	C	C	C
	• Authoritative body	C	E	E	C	E	C	E	E	E	C	E
	• Academic authority	E	C	E	C	E	E	E	E	E	C	E
	• Neutrality across industries, products or processes.	C	C	C	E	C	D	C	C	A	A	A
	• Relevance for current policy	C	C	C	E	C	C	C	C	C	A	C

threat to soil, is globally applicable and spatially explicit. The model does not include an uncertainty assessment but the authors point out that the main uncertainty source comes from the assumption of linearity between SOC and NPP depletion – field in which there is uncertain results from studies on the link between soil erosion and biomass production. Also the simplifications performed to convert the variability of soils to units of NPP or emergy would require further refinement. Model documentation is easily accessible and transparent. As for the model applicability, the model characterization is based, among other variables, on land management and land use, although CFs are not directly associated to those, not allowing for a straightforward application of the model based on land use inventory data. Moreover, land use types considered would require an adaptation to ILCD flows.

***Alvarenga* et al. *(2013)*.** It provides global exergy CFs – based on biomass extracted for natural land and on potential NPP for human-made land. This model has limitations regarding environmental relevance mostly because it gives a poor impact characterization (CFs values are often the same for different land use types). CFs are provided – and directly linked to - only for a limited number of land use flows, and consequently, there is reduced coverage of ILCD land use inventory flows. Some adaptation to the ILCD flows would be as well needed. Conversely, the model has a good spatial resolution and uses an indicator that could potentially distinguish between intensive and extensive uses. Regarding scientific robustness, the model is on average similar to others. It provides an application example that is up-to-date, but it lacks a thorough uncertainty analysis, as only 95% confidence intervals are included - and these are calculated for the aggregation at country level process rather than as an error measure that involves the uncertainty of the original data, whose quality is not assessed. The model is well documented.

***Alvarenga* et al. *(2015)*.** It calculates HANPP, as proxy of many aspects related to soil quality, biotic production and biodiversity. The model allows for a global application since CFs are provided for all countries and by land use type. The model is also capable to distinguish different agricultural land use flows, while it does not distinguish among different types of sealed land, forest and pasture, being therefore limited in terms of impact characterization. The model presents a very simplified measure of uncertainty while being fully up-to-date in terms of modeling developments. The latter is validated by the good quality of the original HANPP data used, whose application in an LCA context is a novelty. The model is also very well documented, and applicable to LCA with a minimum effort, given that the authors made a good attempt to provide global CFs for 162 countries that are readily usable in LCA and matches with some ILCD land use flows.

***Gardi* et al. *(2013)*.** This non-LCA model performs well on the consistency between those indicators that may be used as midpoints and endpoints, as well as on the relation between inventory and midpoints. Although the model has a high spatial resolution (1×1 km grid cells), its geographic coverage is limited to Europe. Due to its fine resolution, the model delivers information relevant related pressures on soil biodiversity at midpoint level. The characterization is done for different types of land use/cover, with a clear distinction between intensive and extensive land management. However, CFs for the application of the model are not available. A direct association of the composite indicator proposed in this model to land use flows would not be possible, since this indicator derives from many other variables – invasive species, contamination – than just land use. Moreover, the coverage of ILCD LCI flows is low, and a considerable effort would be needed to conform to the ILCD nomenclature, due to the amount of available site-specific flows in the model. Conversely, Gardi et al. (2013) is the more complete model in terms of robustness: it provides a comprehensive analysis of uncertainty, and applies data from known datasets that include also this type of assessment. Also value choices are considered robust and derive from a consensus process including experts. For those and other reasons mentioned above, stakeholder's acceptance and relevance for current policy are high.

***Burkhard* et al. *(2012)*.** Some of the indicators adopted in this model could be considered midpoint indicators, while others would be closer to endpoints. The model indicators' set covers all AoPs but without establishing links between midpoint and endpoints – since all indicators are directly and solely derived from land use types. The model does not provide guidance for a possible aggregation of the information contained in the different indicators. The characterization model is not directly accessible neither mathematically reproducible since it adopts a scoring system largely based on expert judgment, and uncertainty assessment is missing. The partially good score of the model in terms of applicability comes from the fact that the model results, are similar to the CFs concept, i.e. they could be easily convertible to CFs, and are directly associated to land use flows, which moreover could be easily adapted to ILCD nomenclature. Yet, currently these values derive from a set of site-specific cases studies, which does not allow for the global application of the model.

3.2. Evaluation the results by criteria

The main insights of models' evaluation for the aspects considered under each evaluation criteria are provided in the following, where we establish comparisons among the different models, against the criteria.

***Completeness of the scope*.** Overall, models were comprehensive considering the coverage of the AoPs and the reference impact pathway. LANCA, Saad et al. (2013) SALCA-SQ and Burkhard et al. (2012) -being multi-indicators -had the most comprehensive coverage of AoPs. Among those, links explicitly made by the models were found only for the AoPs natural resources and natural environment. As for geographic coverage (see Table 4), models generally allow for the global application of the characterization, except for SALCA-SQ –site-specific–, Gardi et al. (2013) –only for Europe–, Garrigues et al. (2013) and Burkhard et al. (2012) – based on local case studies.

***Environmental relevance*.** The models proposed by Brandão and Milà i Canals (2013) and Gardi et al. (2013) had the most relevant impact characterization of the different land use and cover types, including also management practices. The characterization proposed by Alvarenga et al. (2013) was the one with the lowest relevance since it bundles all human-made land use systems into one. The coverage of ILCD inventory flows was limited for all models with the exception of LANCA (Bos et al., 2016). Most models covered only the impacts resulting from land occupation, while transformation impacts were only estimated by: Brandão and Milà i Canals (2013); LANCA and Saad et al. (2013). Finally, the temporal resolution of the models was generally annual, while the spatial resolution spanned various levels (countries, climatic regions, and biomes) (see Table 4).

***Scientific robustness and uncertainty*.** All models were peer-reviewed and stated their value choices, although an explicit, comprehensive list of these choices was not always reported. Only three models (Garrigues et al. (2013), Alvarenga et al. (2015) and Gardi et al. (2013)) were fully up-to-date, with the remaining models only partially up-to-date in terms of latest scientific findings underpinning the models. Most LCA models have been used for running case studies, but none reported to be ready for more general applications. Regarding input data quality tests and uncertainty assessment, only the model proposed by Gardi et al. (2013) explicitly stated to having undergone this, with the

Table 4
Main CFs applicability aspects of the evaluated models, which determine their ability to be globally applicable. The models allowing for a global application have been highlighted with grey background color. Level 4 of land use flows partially incorporates land management practices.

Model	Applicability of the CFs – if available			
	CFs associated to land use flows	Land use flows coverage by the CFs (hierarchical level and compatibility with ILCD flows)	CFs geographic coverage	CFs spatial resolution
Brandão and Milà i Canals (2013)	Yes	- Level 2-3 - Adaptation to ILCD nomenclature required	Global	Regional (climatic regions) and world default
LANCA (Beck et al., 2010; Bos et al., 2016)	Yes	- Level 4 - Compatible ILCD	Global	Country, world default and local (site-specific)
Saad et al. (2013)	Yes	- Level 1 - Compatible ILCD	Global	Regional (biogeographical regions) and world default
SALCA-SQ (Oberholzer et al., 2012)	No	–	Local (specific for Europe)	Local (site-specific)
Núñez et al. (2010)	No	–	Global	Regional (ecoregions)
Garrigues et al. (2013)	No	–	Some crops in some countries	Country
Núñez et al. (2013)	No	–	Global	Local and country
Alvarenga et al. (2013)	Yes	- Level 2-4 - Adaptation to ILCD nomenclature partially required	Global	Higher than country (grid size of 5′ or 10 × 10 km at the Equator), and world default
Alvarenga et al. (2015)	Yes	- Level 2 - Adaptation to ILCD nomenclature partially required	Global	Country and world default
Gardi et al. (2013)	Partly	- Level 1 - Adaptation to ILCD nomenclature partially required	Europe (but easily replicable globally)	Local (grid size 1 × 1 km)
Burkhard et al. (2012)	Yes	- Level 3 - Adaptation to ILCD nomenclature partially required	Local	Local

remaining models very limited in this aspect.

Documentation, transparency and reproducibility. Access to input data and model documentation, its completeness, as well as accessibility to the models' input data were good overall, with spatial data mainly available in a Geographic Information System (GIS) format. Six models had CFs entirely available and in an adequate format for LCA practitioners, while the remaining ones had some limitations in this regard (see results for applicability below). In terms of reproducibility, the most limiting aspect was the availability of the information about the characterization model, and the fact that the model itself was not available in an operational format, i.e., easy to be implemented in LCA software.

Applicability. LANCA in its latest version (Bos et al., 2016) stood out in terms of applicability, with CFs available both as global default and at country scale. Overall, the assessed models have LCI flows available, although not always complete and not corresponding to recommended nomenclature, such as that of Koellner et al., 2013b. For some models —SALCA-SQ, Núñez et al. (2010, 2013), and Gardi et al. (2013)— the inventory data were only partially available, requiring the processing of spatial data, and therefore challenging the applicability factor. The usability of the CFs was the main issue in terms of applicability. CFs for a global application were available for the majority of the models (Table 4). Regarding spatial differentiation, those having CFs at country scale are more easy to be practically implemented in LCA software. On the contrary, models providing CFs only by biomes or climate region would require an additional effort to be incorporated.

A compilation of the CFs for the application of these models using global default values is provided in the Supplementary Material (SM 3). The models by Núñez et al. (2010, 2013) and Garrigues et al. (2013) were based on inventory flows which are very specific and expected to be collected by practitioners. Among the models that allow for a global application, only the CFs for LANCA (Bos et al., 2016) and Saad et al. (2013) followed a land use classification fully compatible with the ILCD; while other three models (Brandão and Milà i Canals, 2013; Alvarenga et al., 2013, 2015) would require (further) adaptation to the ILCD classification system. A total of five models provided world values based on the upscaling of data from country and/or regional scales. In addition, three models were able to characterize the impact at the level of climatic regions, ecoregion or biomes (see Table 4), four at the country level, and five with a local spatial resolution. The compilation of global CF values, when available, is included in the Supplementary Material (SM 3). Graphs comparing these CFs for a selection of representative land use types is available in Supplementary Material SM 4 and SM5 for land occupation and transformation, respectively.

Stakeholders' acceptance. Results for this set of criteria were moderate overall, mostly due to missing authoritative/institutional bodies who are already applying or endorsing the models. The best results were for Gardi et al. (2013). Gardi et al. (2013) was the most relevant model for policy, especially regarding soil policies in the EU (The European Atlas of Soil Biodiversity- Jeffery et al., 2010); while the remaining models could be integrated into the soils protection policies, for example. Neutrality was challenged by some models, mostly due to the limited coverage of land use flows by the impact assessment such as models only addressing agricultural activities/land use types, e.g., SALCA-SQ and Garrigues et al. (2013).

4. Discussion and outlook

The models reviewed in this study are highly heterogeneous, spanning from those providing indicators for specific soil properties (e.g. SOM), to those assessing one or several soil threats (erosion, desertification, etc…), up to those considering the provision of ecosystem services based on soil functions. Presently, no model meets all the features required by the criteria defined in the systematic evaluation developed in this study. For instance, no model entirely combines a relevant characterization of the multiple impacts on soil with a sufficient maturity for being easily applied in an LCA study, i.e., the availability of CFs with a global coverage and directly associated with land use inventory flows. Nevertheless, compared to the evaluation conducted for the current ILCD recommendations (EC-JRC, 2011), the newly developed models appear more robust and improved in terms of the scope completeness and geographical coverage. They are also more data-intensive, but their input data is more accessible, as are CFs and the models themselves. In the following, the key findings of the present study are summarized, representing research needs towards robust modeling of impacts on soil quality.

4.1. Position of the indicator(s) in the impact pathway

Ideally, a midpoint indicator should be a relevant building block for the calculation of the endpoint indicators towards the different AoPs. However, only a few amongst the assessed models explicitly identified the theoretical links between midpoint and endpoint. The only example in the literature is a recent study carried out by Cao et al. (2015), which quantifies the monetary value of ecosystem services, understood as an endpoint indicator based on Saad et al. (2013). Moreover, the AoPs covered by the models often differ, ranging from *natural environment* to *natural resources*, even for studies referring to the same type of indicators.

If the application of a single indicator was desirable, the selected models at midpoint should be a reliable proxy for the relative performance of several specific indicators. In this regard, NPP and HANPP could represent good indicators due to their capability of representing a synthesis of different specific indicators. They may also be used for supporting endpoint modeling, covering two AoPs (*natural environment* and *natural resources*). Theoretically, if soil is lost through erosion, affected by salinization, or its fertility is reduced (e.g., having less SOM available) then the NPP should be low. However, current models that apply NPP in LCA have a very limited coverage of land use elementary flows, and do not capture different land management practices, although the same could be said of other models assessed. Moreover, HANPP and NPP consider the agricultural use of the land as beneficial (negative value of the CF), since this human intervention increases the overall biomass production. This is set without accounting for the extra input needed to obtain the productivity increase, and the additional impacts of agriculture intensification on the quality of the soil and the whole agroecosystem.

4.2. Comprehensiveness and specificity in covering key impacts on soil quality versus applicability

The applicability of the models requires CFs associated to land use inventory flows, whereas specificity entails capability of addressing spatial variability. Good applicability and specificity has been reached by five out of the eleven models (Brandão and Milà I Canals, 2013; LANCA, Saad et al., 2013; and Alvarenga et al., 2013, 2015). In the case of Brandão and Milà I Canals, 2013, the model can be even easily expanded using SOC field observations. For example, Morais et al. (2016) used the LUCASOIL database and this model to calculate land use CFs for the European Union. Regarding the other models, they did not always stand out in terms of environmental relevance and comprehensiveness. The opposite pattern was found for other models, which scored better in terms of comprehensiveness and relevance —and even scientific robustness— while missing a straightforward applicability in LCA (e.g. SALCA-SQ, Gardi et al., 2013). In addition, models dealing with relevant soil threats (Núñez et al., 2010, 2013; Garrigues et al., 2013) were more limited in terms of applicability, requiring numerous data from the inventory or being incompatible with the ILCD land use elementary flows. This shows the frequent trade-offs between comprehensiveness, environmental relevance and applicability and raises the question of if the applicable models are addressing the key impacts on soil.

Additionally, some models addressing relevant aspects of soil quality have not been included in this evaluation since their level of development did not allow for the model application. This includes, for instance, the salinization models evaluated by Payen et al. (2014) — which were assessed by the authors following the ILCD handbook criteria. Salinization is also an important threat to soil: even if taking place only in a limited geographic area high salinity area in a very dry climate could be barren for an infinite time period, leading to a permanent impact (Koellner et al., 2013b). Also, promising models might arise outside from the LCA field, as shown by the models by Burkhard et al. (2012) and Gardi et al. (2013). Further work might allow for the use of this type of models, once applicability limitations have been overcome.

The results of this study serve also to identify additional needs for the complete applicability of models. This includes the calculation of transformation impacts -missing for most models-as well as guidance for the calculation of normalization factors, totally absent from all evaluated models.

4.3. Single indicator versus multi-indicator models

The need of multiple indicators to thoroughly assess soil quality was expressed both by modelers that account for various drivers of impact (LANCA; Saad et al., 2013; SALCA-SQ) as well as pointed out by those using only one indicator (e.g. Garrigues et al., 2013). For example, notwithstanding single indicators such as SOM are considered one of the most important indicators for the sustainability of cropping systems (Fageria, 2012) and plays a crucial role in provisioning and supporting ecosystem services, the capability of SOM to be a good proxy of the overall impacts on soil could be questioned. However, our comparison of the CFs for multi-indicators — available in the Supplementary Material (SM3, 4 and 5) — shows that the information given by some of these indicators could be redundant, which points out several research needs, including: 1) statistical analyses of the redundancy of CF values in multi-indicator models; 2) analysis of the sensitivity of results to using multiple instead of a single indicator. Basically, this means answering the questions: "which is the role of each different indicator in the overall contribution of a land use flow to the total impact? which is the key driver that will determine the ranking of two production options?" Despite multi-indicator models aiming at greater completeness, several relevant soil threats could be still missing (e.g. desertification, salinization). Those threats, so far, have been only addressed by specific models. Moreover, none of the multi-indicator models provides guidance on how to aggregate the different indicators in a single score, which introduces an additional complexity in the analysis. Thus, model developers should either provide guidance on how to assess the relevance of each of the multiple indicators, or propose an aggregation algorithm. Moreover, chosen indicators should provide non-redundant information and facilitate the usability of the models. To this regard, Helin et al. (2014) found a high correlation among some indicators within the set proposed in Saad et al. (2013). They also showed a correlation between those indicators and indicators from other models – e.g. Ecological Footprint and HANPP. When deciding the most adequate indicator(s) among the proposed sets, it should be bore in mind that for many applications of LCA the ultimate target of the quantification of impacts is to compare the performance of different products rather than an accurate calculation of the environmental impact. Also, a coarse scale, the one available for the applicable models, might be not be adequate for indicators that require a very detailed spatial analysis (Koellner et al., 2013a), e.g. erosion.

4.4. Land use management: discriminating land use intensities

The ideal model for the quantification of the impact of land use on soils should be able to differentiate amongst different land use intensities and different management practices. To this regard, the CFs of the globally applicable models did not sufficiently differentiate among diverse types of land use, and hardly reflected different management practices. Even for models that provide CFs for a complete set of land use flows, the discriminating power was poor, with equal/similar characterization values for many land use types simultaneously. The models by Brandão and Milà i Canals (2013), and Gardi et al. (2013) were exceptions to this poor discriminating power of the models towards land management.

Moreover, while appropriate agro-ecological crop and soil management practices – based on the knowledge of ecological processes – may represent a solution for food security and soil sustainability (Baulcombe et al., 2009), those concepts are almost completely neglected by the evaluated models – only partially covered in SALCA-SQ. Consequently, current LCA methods are unable to rank interventions considering if “good” agricultural practices are put in place. This aspect might be included as, for example, qualitative information in the impact models.

In this review, the need for a more comprehensive land use classification was identified, fact that was already previously pointed out by Allacker et al. (2014). Moreover, even the land use classification used here as reference has limitations in reflecting the potential differential impact of diverse land management regimes on the soil indicators. An ideal classification system would enable the assessment the environmental benefits of farming practices, which might have a strong influence in the impact of the production chain, and which is usually not taken into account in LCA studies. Payen et al. (2014) and Helin et al. (2014), comparing some models reached similar conclusions for agricultural and forest land use types respectively. However, in order to develop a comprehensive, yet applicable and sound land use classification, it is important to assess the data limitations encountered both by LCA practitioners and by model developers.

4.5. Nomenclature of elementary flows

An additional need is to push forward a consensus land use classification that is comprehensive and flexible to the availability of data at different levels of detail. One example is the hierarchical classification proposed by Koellner et al. (2013a), which is also based on widely recognized land use classifications (CORINE Land Cover, Global Land Cover). This classification might not be optimal for the specific assessment of soil quality impacts. Yet, using such classification would improve the comparability of results among models, and allow adapting to different spatial coverage and resolution. This specific classification includes also some detail on land management practices (at 3rd and 4th classification hierarchical level) for agriculture and water bodies, although some further detail should have to be added depending on the case study. A further challenge for the nomenclature of the elementary flow as well for the inventories is associated to the possibility of capturing, in future, not only aspects which affect the field but also those related to a greater scale, e.g. at landscape level.

4.6. Data to be collected from the inventory: is area of land use enough?

An additional challenge is the analysis of which type of inventory data would be the most relevant to assess impacts of human disturbances on soil quality. To this regard, building the inventory based on only land use flows is questioned by several authors (Garrigues et al., 2012, 2013, Oberholzer et al. 2012; Helin et al., 2014). For example, Helin et al. (2014) stated that in some cases other type of information would be preferable – e.g. for forest biomass the amount of biomass harvested would be probably better than the amount of land occupation in order to quantify the environmental impact. However, this might introduce confusion between products and inventory data, since e.g. biomass harvested is an output to the technosphere. Other approaches under discussion involve specifying elementary flows associated with changes in soil quality directly in the inventory. For instance, flows associated with net acidification rate can be explicitly calculated in the inventory (and are not spatially dependent) by using data on the ash alkalinity of exported product combined with the acidifying effect of different fertilizers (expressed as an elementary flow of hydrogen ions to soil water). Both yield of product and fertilizer inputs are both explicitly quantified inventory flows and a direct mathematical relationship can be established between them and soil acidification rate.

Although we can agree that inventory flows based on land use do not provide the full picture of the impact, it is important to bear in mind that making inventories complicated to create (including e.g. data on soil conditions, climate, or location) could hamper LCA practice, especially where different soil quality attributes vary in importance across different locations. We are facing a complex system with a lot of interactions and specific features for which it is difficult to balance between comprehensive and feasible data. Even the assumption that soil quality changes derive solely from changes in land use/cover and keep constant during land occupation is too simplistic to grasp the complexity of the interactions taking place at the soil system. However, ideally, assessing impacts on soil quality requires the knowledge of at least the location of the production system, in order to consider the local environmental conditions (e.g. climate), altogether with a clear definition of the flows associated to different management practices.

The need to consider these aspects is still challenged by the stringent structure of the LCIA framework, which – although enables easier applicability of LCA software - very often imposes limitations to modeling. It is therefore, clear that the scientific community should further explore approaches as 1) the use of substance flows that can be easily calculated in the inventory, and which are not spatially explicit, and associate this information with soil properties and functions; 2) the development of more detailed archetypes for land use that capture the impact of different management interventions (with appropriate CF for the different archetypes); 3) promote the use of GIS software in LCA, in order to better depict spatially explicit information both in inventory and LCIA. The information related to soil quality could be also directly incorporated into life cycle inventories.

4.7. Land use, climate change and resource depletion: sharing elements of the impact pathways

Having a clear target for the desired endpoint is an essential aspect when selecting the midpoint impact models for soil quality. There is thus an urgent need for defining a consensus land use impact pathway. It is important to align advances in the cause-effect chain of land use involving soils and e.g. biodiversity. The land use impact pathway should be also consistent with the cause-effect chain determined by other impact pathways where soil also plays a role – climate change, land use as a resource. This would be likely to reduce the risk of double counting the impacts derived from land use.

4.8. Uncertainties

Our evaluation included a preliminary screening analysis of the CFs, whose associated uncertainties were assessed by only few of the evaluated studies (Gardi et al., 2013 stood out in this regard). Therefore, a further quantitative assessment of the models prior to any possible recommendation of their application by LCA practitioners it is strongly advisable. Such an assessment should also include the analysis of the spatial variability of CFs, and the relative relevance of the impacts resulting from land occupation and transformation – CFs not provided by most models. This relative relevance might derive from the underlying modeling choices – e.g., time horizon choice–; but they might also be case study dependent (Helin et al., 2014; Mattila et al., 2012), therefore reflecting the real differential impact of the occupation and the conversion of land caused by different production chains.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2016.05.077.

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