European Commission

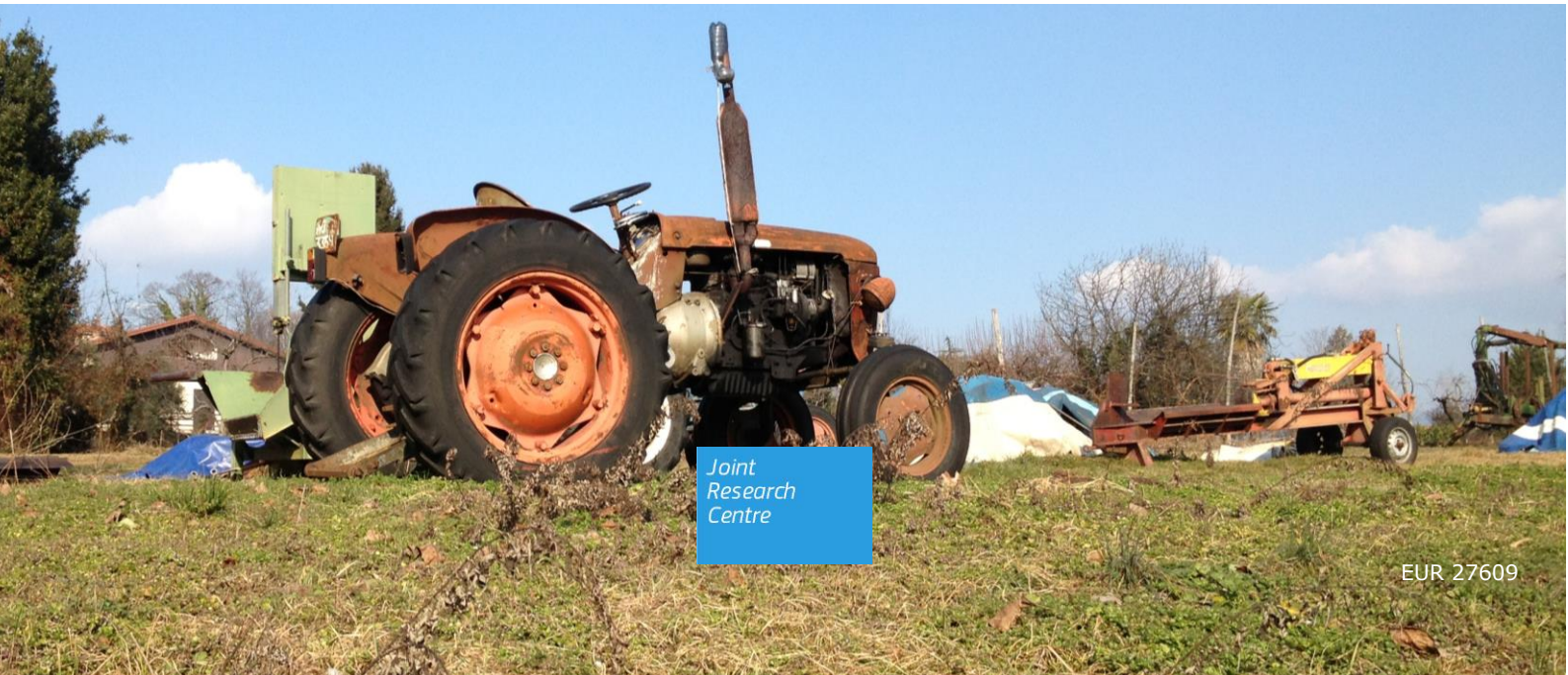
JRC TECHNICAL REPORTS

Land-use related environmental indicators for Life Cycle Assessment

Analysis of key aspects in land use modelling

Beatriz Vidal-Legaz, Serenella Sala, Assumpció Antón, Danielle Maia De Souza, Marco Nocita, Ben Putman, Ricardo F.M. Teixeira

2016



EUR 27609

Contact information
Name: Serenella Sala
Address: Via Enrico Fermi, 2749 ISPRA (VA) Italy
E-mail: serenella.sala@jrc.ec.europa.eu

JRC Science Hub

https://ec.europa.eu/jrc

JRC 98694

EUR 27609

ISBN 978-92-79-54023-3 (PDF)

ISSN 1831-9424 (online)

doi: 10.2788/905478 (online)

Luxembourg: Publications Office of the European Union, 2016

How to cite the report:

Vidal-Legaz B, Sala S, Antón A, Maia De Souza D, Nocita M, Putman B, Teixeira RFM (2016) Land-use related environmental indicators for Life Cycle Assessment. JRC Technical report. Luxembourg: Publications Office of the European Union, pp.44 doi: 10.2788/905478

Table of contents

Acknowledgments

The work on assessing land use models was partly financially supported by the Directorate General Environment (DG ENV) of the European Commission in the context of the Administrative Arrangement 'Technical support for Environmental Footprinting, material efficiency in product policy and the European Platform on LCA' (No. 07.0307/ENV/2013/SI2.668694/A1). R.F.M. Teixeira was supported by the grant SFRH/BPD/111730/2015 from Fundação para a Ciência e Tecnologia. The authors would also like to thank Luca Zampori for the review of the document. We are also very grateful for the support of some model developers, which enabled a profound understanding of their models: Hansrudolf Oberholzer, Ulrike Bos, Monserrat Núñez, Cecile Bulle and Rodrigo A.F. Alvarenga.

Abstract

Soil is a key environmental compartment, determining the supply of crucial ecosystem services as well as supporting biodiversity below and above ground. In the last 15 years, substantial efforts have been made to incorporate the impacts on land due to production processes - from a life cycle perspective-, in order to improve the comparison of the environmental performance of products. The use of robust models that enable evaluating the effects of land use interventions on soils is critical for obtaining meaningful Life Cycle Assessment (LCA) results. In order to support the recommendations of adequate models, this study:

- Reviews and compares a set of models for relating land occupation and land transformation to soil indicators at midpoint level, addressing soil properties and functions as well as threats to soil
- Proposes a new land use cause-effect chain to appraise the impacts of land use interventions on soils
- Analyses the characterizations factors provided by the reviewed models

The comparison is done through a systematic evaluation, based on the International Life Cycle Data system (ILCD) set of criteria, which considers aspects such as the scientific soundness of the models, their ease of applicability for LCA practitioners, and their level of acceptance by stakeholders.

The study is complemented by a supplementary material which contains: the list of the evaluation criteria, the filled-in evaluations and the compilation of characterization factors for the evaluated models that allow for a global application. The supplementary material could be downloaded from http://eplca.jrc.ec.europa.eu/uploads/supplementary-material-land-use.xlsx

Based on the results of this evaluation, this study identifies valuable approaches and highlights the strengths and limitations of the reviewed models. Research needs for improving the assessment of land use impact on soil in an LCA context are also identified, namely: 1. adopting a common land use cause-effect chain and land use classification; 2. accounting for different land management practices; 3. assessing the added value of using multi-indicators, as some evaluated models propose, for a comprehensive account of impacts of land use on soils; 4. ensuring consistency from midpoint to endpoint indicators; 5. providing guidance to calculate normalization factors; and 6. ensuring a systematic assessment of models results uncertainty.

1. Introduction

Life Cycle Assessment (LCA) is a valuable approach to quantify the potential impacts of the production chain from a life cycle perspective (Hellweg and Milà i Canals, 2014). In this context, the impact of products on the land is a crucial component to be considered by LCA studies.

Impact assessment in an LCA context requires the use of reliable models. To this aim, the European Commission International Reference Life Cycle Data System handbook (ILCD handbook) underwent an evaluation of possible models that could be used for a robust estimation of the different typologies of impacts along the production chain, including also the impacts associated to the occupation and transformation of land (EC-JRC, 2011 and 2012). This evaluation, which targeted models developed until 2009, assessing them against a defined set of criteria including, among other, models' environmental relevance, applicability, robustness and stakeholders' acceptance. Based on this evaluation, some models were recommended for the different impact categories. Yet, land use impact assessment models did not fully meet the quality requirements detailed by the evaluation criteria, and only one model was recommended, the model by Milà i Canals et al. (2007a), which uses Soil Organic Matter (SOM) change as stand-alone indicator to approach the impacts derived from land occupation and land transformation. The model was considered applicable with caution at midpoint[1] level, while none of the evaluated models was recommended at endpoint level.

Over the years, several land use-related impact assessment models have been developed. Hence, a review of methods has been undertaken in this study with the aim of assessing progress in land use impact modelling and proposing the way forward, with specific regard to application in the context of the Environmental Footprint (EF) (EC, 2013). This report represents a background document, reflecting the analysis of the models needed for building the paper of Vidal Legaz et al. (2016).

1.1. LCIA land use models: state-of-the-art

In the last 15 years, substantial efforts have been made to improve the assessment of the impacts on land use derived from production supply chains. This includes the impact of both land interventions, i.e. occupation and conversion of land – the latter referred to as *transformation* in a LCA context. Yet, a consensus on the best available model for land use has not been achieved (Teixeira et al., 2015).

Currently, many LCIA methodologies assess the impact of land use on biodiversity (endpoint), while midpoint indicators often include just the area of land being occupied and/or transformed for the functional unit (EC-JRC, 2011). 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 United Nations Environmental Programme—Society of Environmental Toxicology and Chemistry (UNEP-SETAC) Life Cycle Initiative task force on land use impacts on biodiversity (Curran et al., 2016). Furthermore, due to the challenges of quantifying the impact derived from land interventions (Li, 2007), soil properties and functions have been

[1] In LCA, the potential impacts associated to the supply chain can be assessed by two types of indicators. On 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.

incorporated in a very limited way. An example is the model by Milà i Canals et al. (2007a) recommended in the ILCD handbook (EC-JRC, 2011), which yet disregards important soil functions systems. Therefore, the assessment of land use impacts needs to be more inclusive (Koellner et al., 2013a). Indeed, according to the UNEP- SETAC Life Cycle Initiative, land use models should focus on soil quality, biotic production, and biodiversity.

An additional challenge for a robust assessment of land use impacts includes also the lack of a clear and consistent cause-effect chain, also called *impact pathway*, which should depict systematically the causal relationships from the inventory data (most commonly amount and typology of land use) to the mid- and endpoint indicators and areas of protection (AoPs). To this regard, Jolliet et al. (2014) pointed out that in order to better understand the link between land use and the provision of ecosystem services and biodiversity loss, we first need to assess the impact of land occupation and transformation on soil quality and functions, on which these services rely. Thus, within the impact pathway proposed for land use, the latter study includes a set of indicators to determine the impacts on soil quality (e.g. soil fertility, soil stability) and the impacts on habitats (e.g. fragmentation, degradation) which lead to the loss of soil and ecosystems. Although this is a valuable step in recognizing the relevance of soil, impacts addressing soil functions are still under-characterized and an in-depth understanding of the connections between mid and endpoint indicators is still required.

1.2. Soil quality and soil functions in LCIA

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, maintain or enhance water and air quality, and 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 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). Therefore, 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).

Given the complexity of the soil matrix, the assessment of soil quality is very challenging. In particular, 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 properties (Milà i Canals et al., 2007b; Garrigues et al., 2012).

The first efforts to address impacts on soil properties and functions in LCIA date back to the 90's, when Heijungs et al. (1997) proposed an evaluation of land use based on the occupation of land surface area by a production system. This first attempt, however, disregarded the state of the soil and its supporting services, such as habitat provision, and properties that frequently depend non-linearly on the spatial and time scale of the intervention. Following, Cowell and Clift (1997; 2000) suggested the first model to assess 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 in three midpoint impact categories (soil erosion, compaction and change in organic matter). The factors were i) mass of soil, ii) living organisms (weeds/weed seeds and pathogens), iii) trace substances (nutrients, salts and pH of soil), iv) non-living matter (organic matter), and v) soil physical properties (texture and structure). A different set of indicators was proposed to evaluate the sustainability of the production capacity of soil (Mattsson et al., 2000) in three different types of vegetable oil crops (rape seed, oil palm and soybean), and in three distinct regions (Sweden, Malaysia and Brazil, respectively). This model considered i) soil erosion, ii) hydrology effects, iii) soil organic matter, iv) soil structure, v) soil pH, vi) accumulation of heavy metals, and vii) phosphorus and potassium content, to assess the soil quality. In addition, the value of soil for nature was addressed by Weidema and Lindeijer (2001), who

proposed five indicators: i) substance and energy cycles, ii) productivity, iii) biodiversity, iv) cultural value, and v) migration and population dispersal. In their proposed impact pathway, they included "altered soil functions" as midpoint category, linking to endpoint life-support functions (e.g. topsoil formation).

Currently, the ILCD handbook recommends the use of the model by Milà i Canals et al. (2007a), which proposes soil organic matter (SOM) as a stand-alone indicator for life support functions. However, although SOM has a crucial role in provisioning (e.g. biotic production) and supporting services (e.g. climate regulation), important soil functions, such as resistance to erosion, or threats to soil such as compaction and salinization, are disregarded (Mattila et al. 2011).

Alternatively, to the use of a single indicator, there is a widespread interest around the need for a minimum set of soil indicators. Thus, 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; Oberholzer et al., 2012). Models following a more qualitative and rather holistic approach in characterizing soil quality have been also developed. Examples of the latter are the multi-indicator model SALCA-SQ (Oberholzer et al., 2012) based on a qualitative scoring system, which is very detailed and comprehensive regarding the list of soil aspects accounted for; and the hemeroby index (Brentrup et al., 2002; refined by Fehrenbach et al., 2015), being hemeroby a measure of naturalness of the land, i.e. calculates the magnitude of deviation from the potential natural vegetation.

The incorporation of threats[2] to soil in the models has been also recently approached. Indeed, focusing on LCA studies, Garrigues et al. (2012) state that models should incorporate what they consider as the main threats to soil and 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 (Garrigues et al., 2012) – e.g. the production of an agricultural commodity could be more easily associated to the erosion caused in a specific type of soil than to a change of soil texture. However, current models that could be applicable in LCA are unable to comprehensively depict the multiple impacts derived on soil derived from land use and land use change.

1.3. Purpose of this study

This study reviews the models that assess potential land use impacts on soils at midpoint level. Specifically, this study builds and expands the review done by EC-JRC (2011) and qualitatively assesses a selection of land use impact characterisation models. A systematic framework based on the existing ILCD handbook, but with revised evaluation criteria, is used for models evaluation. The land use impact pathway has been revisited, and a new version, used as reference in the models' evaluation process in this study, is proposed.

In the models' evaluation, a strong emphasis was given to the immediate applicability. Therefore, aspects such as the availability of characterisation factors (CFs), were central. To this regard, whenever available, CFs of the evaluated models have been compiled and are provided within the models' description. The study is organized as follows:

- methods (section 2), which includes:
    - the selection of models for evaluation (section 2.1);
    - the review of the impact pathway (section 2.2);
    - the criteria for the evaluation of models (section 2.3)

[2] 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.

- results (section 3), which include:
    - the description of models selected for evaluation (section 3.1);
    - the revised land use impact pathway (section 3.2);
    - the evaluation results (section 3.3), provided firsts by model and then by evaluation criteria
- discussion and outlook (section 4)

2. Methods

2.1. Selection of models

In this study, land-use models previously evaluated in the ILCD handbook (EC-JRC, 2011) were re-examined, i.e. it was investigated if relevant new developments have been introduced for these models that would allow for the modeling of land use impact at 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.

The literature review was carried out using the following search query in *Science Direct* until October 2015: "Soil quality/properties/functions/threats + Life Cycle Assessment" and "Soil quality/properties/functions/threats + Assessment/Evaluation", searching for studies published between year 2009 and the present.

The review identified thirty-one models, which were shortlisted for further evaluation only if they: i) computed indicators for assessing soil properties/functions/threats; ii) were compatible with LCA, i.e., they were used to calculate impact indicators starting from elementary flows presented in Life Cycle Inventory (LCI); iii) produced an output consisting of CFs, i.e., the parameters that allow deriving impact (midpoint/endpoint) indicators from the LCI data, or that could be easily converted into CFs. A total of eleven models fulfilled these requirements and further underwent the evaluation procedure, whose description is provided in section 3.1. In addition, other three reference land use-related models are described in section 3.2, and their CF's are reported in supplementary material (SM3). The inclusion of the following models allows a more complete comparison between models.

2.2. Review of the impact pathway

The LCA land use impact pathway for land use was revisited in order to count on a reference pathway for the models' evaluations. It served also for 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., 2007b; 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.

2.3. Criteria for the evaluation of models

An updated version of the ILCD evaluation criteria form was developed to evaluate the models building on those defined in the ILCD handbook (EC-JRC, 2011) and in Curran et al. (2016). The set of criteria includes a descriptive section (*summary information*), followed by an evaluation section in which models are qualitatively assessed against: i)

completeness of the scope, ii) *environmental relevance*, iii) *scientific robustness and uncertainty*, iv) *documentation, transparency, and reproducibility*, v) *applicability*, and vi) *stakeholders' acceptance*. Scores between A and E were assigned, which generally mean, respectively, the best (A) and the worst (E) possible results. Among the full set of criteria, land use-specific criteria are to be found under *environmental relevance.*

The full set of evaluation criteria is listed in Table 1, while the supplementary material (SM1) includes the complete form with models' evaluations, as well as details on the scoring rules. A brief description of each evaluation criteria is provided next.

Summary information

For each model we compiled relevant background information, which helped carrying out the evaluation: 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, using the ILCD flows list (October 2015, http://eplca.jrc.ec.europa.eu/?page_id=140) as reference.

Completeness of the scope

Under this criterion, we assessed if the proposed midpoint indicator(s) cover a relevant gap in the information needed to consistently compute the endpoint indicator, as well as how close is the link of midpoint indicators to inventory flows. Last, we determined which areas of protection (AoPs) are covered, and evaluated the geographic coverage (local/regional/global) of each model.

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 the land use classification proposed by Koellner et al. (2013a). This ILCD land use classification includes a complete set of land use types, and aggregates them in four hierarchical levels. This criterion also assesses if the model addresses land use's specific aspects: i) the distinction between extensive and intensive land uses; ii) whether the model considers the impacts of both land occupation and transformation; and iii) the type of soil indicators included, e.g., soil properties, functions, and threats. Finally, the spatial-temporal resolution of the model input data was evaluated as a determinant aspect of the accuracy of the model results.

Scientific robustness and uncertainty

Under scientific robustness, criteria are included to assess if there is evidence of the model soundness –whether it has been peer-reviewed, the robustness of model choices and the use of up-to-date knowledge. In addition, we substantiated the existence of case studies that confirm the applicability of the model. Under uncertainty, we detailed whether quality checks of the input data have been carried out, and if figures to report uncertainty are provided.

Documentation, transparency and reproducibility

The accessibility and the completeness of the model documentation was assessed, together with the accessibility of LCA practitioners to the input data and CFs – whether they are available and in which format. The accessibility of the characterisation model itself and of the modelling choices was assessed as well.

Applicability

As models may be scientifically relevant and comprehensive but difficult to implement in practice in an LCA study, the compatibility of the model with the scope of the LCA framework was assessed together with some other practical aspects, such as i) availability of the inventory flows datasets required for the application of the model; ii) level of

implementation of the CFs in the LCA software; iii) availability of normalization factors; and iv) ease to conform to the ILCD classification. We assessed the spatial-temporal resolution of the CFs, and the spatial resolution of the inventory flows.

Stakeholders' acceptance

Under this criterion the understandability of model results and its associated uncertainty was evaluated. Also, it was specified whether the model has an academic and/or authoritative body behind that increases its reliability by the stakeholders, and whether the model is neutral, i.e. if it does not favour a specific sector or industry. Finally, it was assessed whether model produces an output relevant to policy.

Table 1: Criteria for land use models' evaluation. See supplementary material (SM1) for details.

Description		Aspect
Summary information (descriptive)		• Purpose and interpretation of the characterisation model • Description of the impact pathway of the characterisation model • Impact categories covered by the midpoint characterisation model • Midpoint indicators • Reference state and time frame • Underlying model(s) • LCI flows covered by the characterisation model
Evaluation criteria		**Sub-criteria**
Completeness of scope		• Impact pathway consistency between mid- and endpoint • Midpoint indicator placement in the impact pathway regarding LCI flows • AoP coverage by the midpoint characterisation model • Geographic coverage
Environmental relevance	Comprehensiveness	• Coverage of ILCD LCI flows • Distinction extensive/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 input • Spatial resolution of the input
Scientific robustness & Uncertainty	Scientific robustness	• Characterisation model peer review and (peers) acceptance • Value choices robustness • The model reflects the 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 characterisation model documentation • Completeness of the characterisation model documentation • Accessibility of the input data • Accessibility of the characterisation factors • Accessibility of the characterisation model • Modelling assumptions and value choices
Applicability		• Compatibility with LCA methodology • Availability of LCI datasets for the application of the model • Usability of characterisation factors for LCA practitioners • Availability of normalization factors for LCA practitioners • Ease to update to conform with the ELCD nomenclature and units • Temporal resolution of characterisation factors • Spatial resolution of characterisation 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

3. Results

3.1. Description of models selected for evaluation

Eleven models were selected and further evaluated against the criteria described in section 2.3, which are briefly described in the following. Within the models' description, the CFs at global level associated to land use interventions, when provided, are also displayed. An overview of the models evaluated and the main features of their CFs is provided in Table 2. Furthermore, the compilation of the world CFs of these models adapted to the ILCD land use flows is provided in the supplementary material (SM3). In addition, three additional models are described in section 3.2 to provide a more complete overview of land –use related models. This includes the current ILCD Handbook recommendation; the further development of one of the evaluated models; and a reference land use endpoint model.

Table 2: Models evaluated in this study and CFs description.

The models allowing for a global application have been highlighted in bold and with grey background color. Level 4 of land use flows partially incorporates land management practices.

Model	Main indicators	CFs				
		Availability (or guidance for CFs calculation)	**ILCD compatibility** of land use flows*	**Land use flows**	**Geographic coverage**	**Spatial resolution**
Brandão & Milà i Canals (2013)	-Soil Organic Carbon (SOC) –as indicator of Biotic Production Potential (BPP)	CFs associated to land use flows	Adaptation required	Level 2-3	Global	World and climate regions
LANCA - Land Use Indicator Value Calculation in Life Cycle Assessment (Beck et al., 2010 (model); Bos et al., 2016 (CFs))	-Erosion resistance -Mechanical filtration -Physicochemical filtration -Groundwater replenishment -Biotic production	CFs associated to land use flows	Yes	Level 4	Global	World, country, and local (site-specific application)
Saad et al. (2013)	-Erosion resistance -Mechanical filtration -Physicochemical filtration -Groundwater recharge	CFs associated to land use flows	Yes	Level 1	Global	World and biogeographical regions
SALCA-SQ - Swiss Agricultural Life Cycle Assessment for Soil Quality 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	CFs not provided. Guidance for site-specific implementation	-	-	-	Local (site-specific application)
Nuñez et al. (2010)	-Desertification index	CFs not associated to land use flows. Guidance for site-specific implementation	-	-	Global	Ecological regions**
Garrigues et al. (2013)	-Total soil area compacted -Loss of pore volume	CFs from site-specific case studies. Guidance for site-specific implementation	-	-	Local (some crops in three countries)	Local (site-specific application)
Nuñez et al. (2013)	-Emergy -Net Primary Production (NPP) depletion	CFs not associated to land use flows	-	-	Global	Regions** and country**/***
Alvarenga et al. (2013)	-Exergy of natural land (biomass extraction-based) -Exergy of human-made land (potential NPP-based)	CFs associated to land use flows	Adaptation required	Level 2-4	Global	World and country** (based on grid size of 5′ or 10×10 km at the Equator)
Alvarenga et al. (2015)	-Human Appropriation of NPP (HANPP)	CFs associated to land use flows	Adaptation required	Level 2	Global	World and country
Gardi et al. (2013)	-Soil pressure (on biodiversity)	CFs not provided	-	Level 1	Europe	Local (grid size 1x1 km)
Burkhard et al. (2012)	-Ecosystem integrity indicators (7) -Ecosystem services indicators (22) -Demand of ecosystem services (22)	CFs not provided	-	Level 3	Local	Local (site-specific case studies)

* If CFs are provided and associated to land use flows
** Not associated to land use flows
*** Based on further development by the authors, provided in October 2015.

Brandão and Milà i Canals (2013)

It is an updated version of the model by Milà i Canals et al. (2007a), which was recommended in the ILCD handbook, and includes Soil Organic Carbon (SOC) as stand-alone soil quality indicator. SOC was considered by the authors the best indicator to approach the biotic production capacity of the soil, which will in turn affect the *natural resources* and *natural environment* AoPs.

This study provides CFs for *occupation* (Figure 1) and *transformation* (Figure 2) for eight land use types (six agricultural typologies, grassland and sealed land) and under different cropland management practices. Unlike the model by Milà i Canals et al. (2007a), which characterized impacts only in the United Kingdom, Brandão and Milà i Canals (2013) provide world values CFs and CFs by climate region (spatially –resolved CFs values not displayed here).

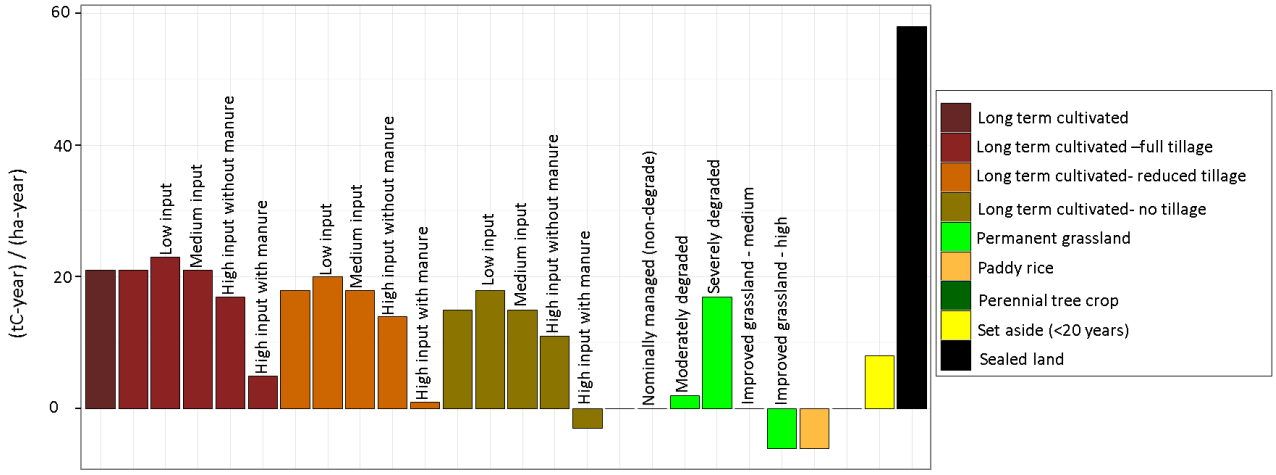


Figure 1: World CFs for land occupation by land use type calculated by Brandão and Mila i Canals (2013).
The highest impacts on SOC are found for sealed land, followed by long-term cultivated land under full tillage regime. Within long-term cultivated land uses, differential CFs values are assigned depending on the level of inputs and manure, being the occupation with high input with manure the one with the lowest impact. For grassland, CFs for different levels of degradation are given. Land use classes are referred to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories.

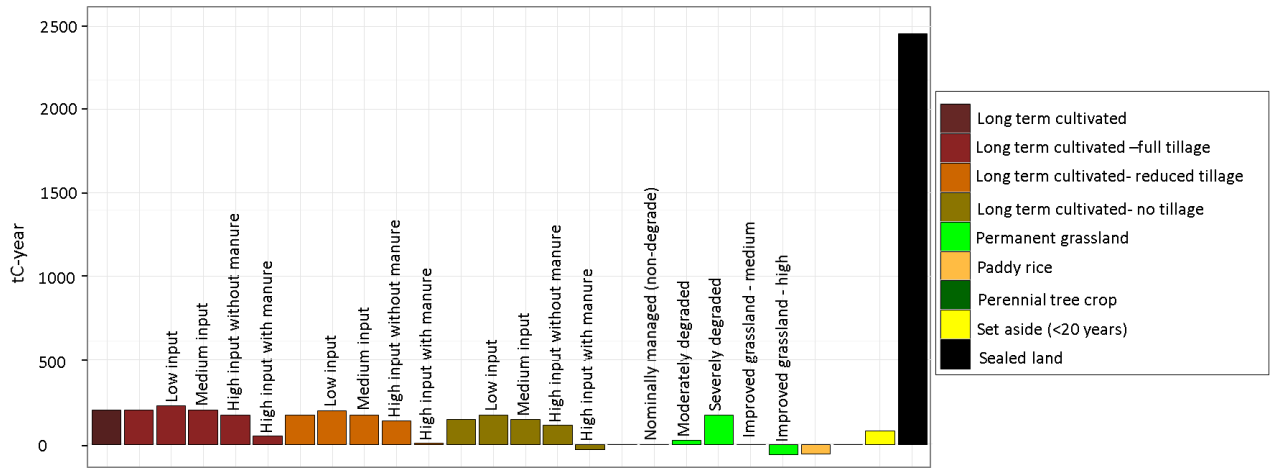


Figure 2: World CFs for land *transformation to* by land use type calculated by Brandao and Mila i Canals (2013).
Same values with negative sign correspond to transformation from. CFs for land transformation follow a similar pattern than for land occupation, but showing a more remarkable difference between the impact of sealed land and the remaining land use typologies.

LANCA (Beck et al., 2010)

It is an updated version of the model by Baitz (2000), which was already evaluated – although not recommended – in the ILCD handbook, partly because of the lack of CFs and the high level of input data requirements. It calculates the following indicators of soil functions: erosion resistance (ER), mechanical filtration (MF), physicochemical filtration (PF), groundwater replenishment (GR) and biotic production (BP).

The model does not establish a specific link to endpoint indicators or AoPs and requires very complete (and site-specific) inventory data, including also land management practices. Although the model by Beck et al. (2010) was originally used to calculate the impact associated to a set of mining and agricultural processes for a selection of countries (both for occupation and transformation), LANCA developers have recently calculated CFs directly associated to land use flows (Bos et al., 2016), which allow for the global application of the model (Figure 3)Figure 1. These include world values and country level CFs (not displayed here) for all ILCD land use flows. Yet, CFs values for e.g. different agricultural land use types or different forest types do not differ much, especially for some indicators. It is to be noted also that CFs for *occupation* and *transformation to* have exactly the same values.

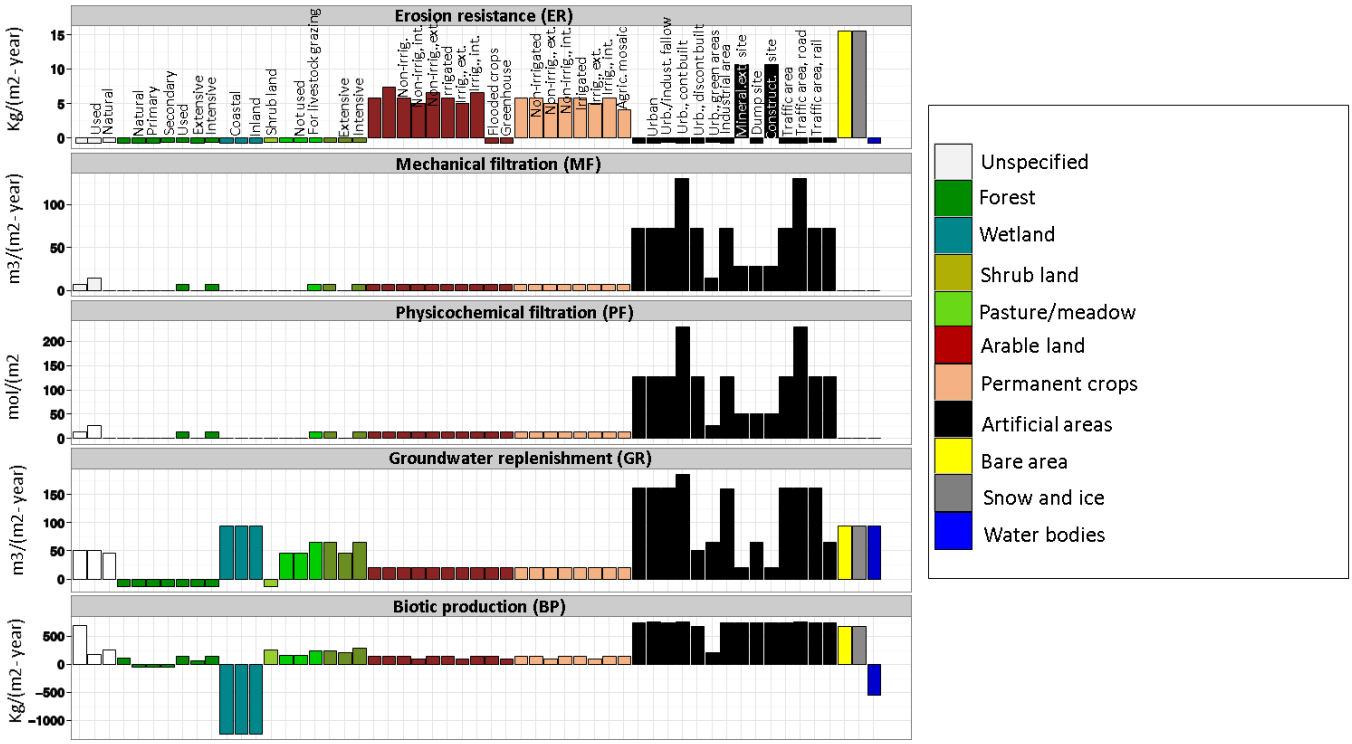


Figure 3: World CFs for land occupation by land use type calculated by LANCA (Bos et al., 2016). The same values as the CFs displayed in Figure 3 correspond to transformation to, and the same values adding a negative sign correspond to transformation from. Similar patterns of impact are observed for all indicators, especially between MF and PF. Artificial land use types are responsible for the highest impact for all indicators except for ER, for which *bare areas* and *snow and ice* surfaces pose a stronger impact, and for which the impact of agricultural land uses is also relatively high.

Saad et al. (2013)

This study develops a global application of LANCA, including some minor methodological modifications and computing all LANCA indicators[3] except for biotic production: erosion resistance (ER), mechanical filtration (MF), physicochemical filtration (PF) and groundwater recharge (GR).

This study calculates CFs directly associated to land use inventory flows and provides CFs for (seven) land use types at high hierarchical level at world level and by biogeographic region (for three different regionalization scales: Holdridge life regions, Holdridge life zones, and terrestrial biomes). CFs are provided both for occupation (Figure 4) and transformation. The authors did not specify whether land transformation refers to *transformation to* or *transformation from*, yet the values reflect *transformation to* CFs.

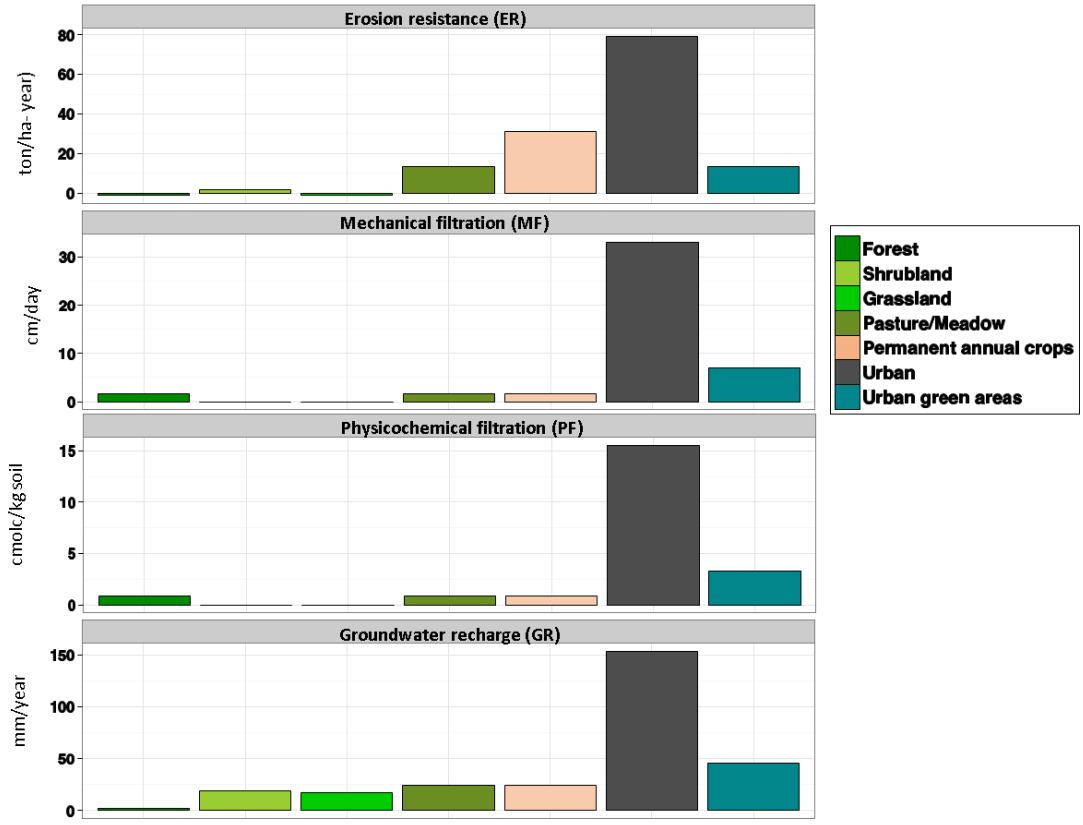


Figure 4: World CFs for land *occupation* by land use type calculated by Saad et al. (2013). Urban land classes showed the highest CFs values for all indicators. Negative CFs values (which means a positive impact) are found for *forest* and *grassland* for the ER indicator. *Shrubland* and *grassland* showed the lowest CFs value for the PF and MF indicators. Finally, *forest* was the land use type that showed the lowest impact on the GR indicator. The CFs for *transformation* show a similar pattern to land occupation, but with a more remarkable difference between the impact posed by *urban* land and the remaining categories.

SALCA-SQ (Oberholzer et al., 2012)

SALCA-SQ is a very detailed multi-indicator model that focuses on soil properties indicators (e.g. macropore volume, microbial activity). The model also uses indicators of threats to soil, such as risk of erosion and compaction. Similarly to LANCA, it does not establish specific links to endpoint indicators or AoPs. It requires very complete (and site-specific)

[3] With some modification in the indicators' naming.

inventory data, including also land management practices, and does not provide CFs for the application of the model at e.g. global or national level.

Nuñez et al. (2010)

Together with Nuñez et al. (2013) and Garrigues et al. (2013) (see just below), this is one of the three models with a focus on soil threats. It calculates a desertification index based on aridity (given by climate conditions), erosion, aquifer over-exploitation and fire risk. This index serves to assess the capacity of the soil to provide ecosystem services and support biodiversity, although these links are not explicitly addressed by the authors.

This study provides CFs for the fire risk index and the variables used in its computation, which cover the whole world by ecoregion (see Table 3). However, CFs are not linked to land use inventory flows (occupation and transformation).

Table 3: CFs of the desertification risk calculated by Nuñez et al. (2010) (dimensionless).

	Marine	Prairie	Temperate steppe	Temperate desert	Savanna	Mediterranean	Tropical/subtropical steppe	Tropical/ subtropical desert
$CF_{Aridity}$	1	1	2	2	1	1	2	3
$CF_{Erosion}$	1	1	1	1	2	2	1	2
$CF_{Aquifer\ overexploitation}$	1	1	1	1.3	1	1.3	1.3	1.6
$CF_{Fire\ risk}$	1	1	1	1	2	2	2	1
CF^{i}	4.0	4.0	5.0	5.3	6.0	6.3	6.3	7.6

i ecoregion

Garrigues et al. (2013)

This model focuses on the impact of soil compaction on topsoil and subsoil, as a result of the use of agricultural machinery. In the model, soil quality is associated to soil compaction, requiring soil data such as texture and water content. However, the model is meant to be part of a broader framework, which should include other processes, such as erosion, change in SOM and salinization, to adequately represent soil quality.

The study provides CFs only from case studies for a limited set of crops and a selection of countries (see Table 4).

Table 4: CFs of soil compaction (total soil area compacted and loss of pore volume) calculated by Garrigues et al. (2013).

Crop		Unit	Maize	Wheat	Triticale	Barley	Pea	Canola	Soya	Sugarcane
Origin			Brittany, France	Brittany, France	Brittany, France	Brittany, France	Brittany, France	Brittany, France	Santa Catarina, Brazil	Punjab, Pakistan
Soil texture			Loam	Loam	Loam	Loam	Loam	Loam	Clay	Loam
Tillage practice			Tillage	Tillage	Tillage	Tillage	Tillage	Tillage	Reduced tillage	Tillage
Total soil area compacted		m^2/ha	19,620	20,690	20,690	20,690	20,036	20,114	15,976	11,566
Loss of pore volume	Topsoil	m^3/ha crop	138.9	128.2	128.2	128.2	135.2	137.5	96.9	95.4
	Subsoil		46.3	43.2	43.2	43.2	43.2	40.8	40.1	32
	Total		185.2	171.4	171.4	171.4	178.4	178.3	137.0	127.4
Yield (dry matter)		t/ha	9.0	7.0	7.0	6.5	4.2	3.3	2.8	35.0
Total soil area compacted		m^2/t crop	2,180	2,956	2,956	3,188	4,805	6,095	5,706	901
Loss of pore volume	Topsoil	m^3/t crop	15.4	18.3	18.3	19.7	32.2	41.7	34.6	2.7
	Subsoil		5.1	6.2	6.2	6.6	10.3	12.4	14.3	0.9
	Total		20.6	24.5	24.5	26.4	42.5	54.0	48.9	3.6

Nuñez et al. (2013)

This study computes the loss of Net Primary Production (NPP) and emergy, as indicators of the damage caused by soil loss to, respectively, ecosystems and resources. Both indicators are based on soil loss calculation through the application of the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978). First, NPP depletion is calculated as a function of SOC loss, which in turn is related to soil loss, and which affects the ecosystem quality. Furthermore, the depletion of soil as resource itself is derived from soil loss by means of an emergy indicator, which expresses all the energy embodied in the system.

This study provides CFs covering the whole world (Figure 5) and CFs for a set of site-specific applications in watersheds in Spain. Recently, CFs at country level have been also calculated by the authors – not yet published. However, neither of these CFs is linked to land use inventory flows (occupation and transformation).

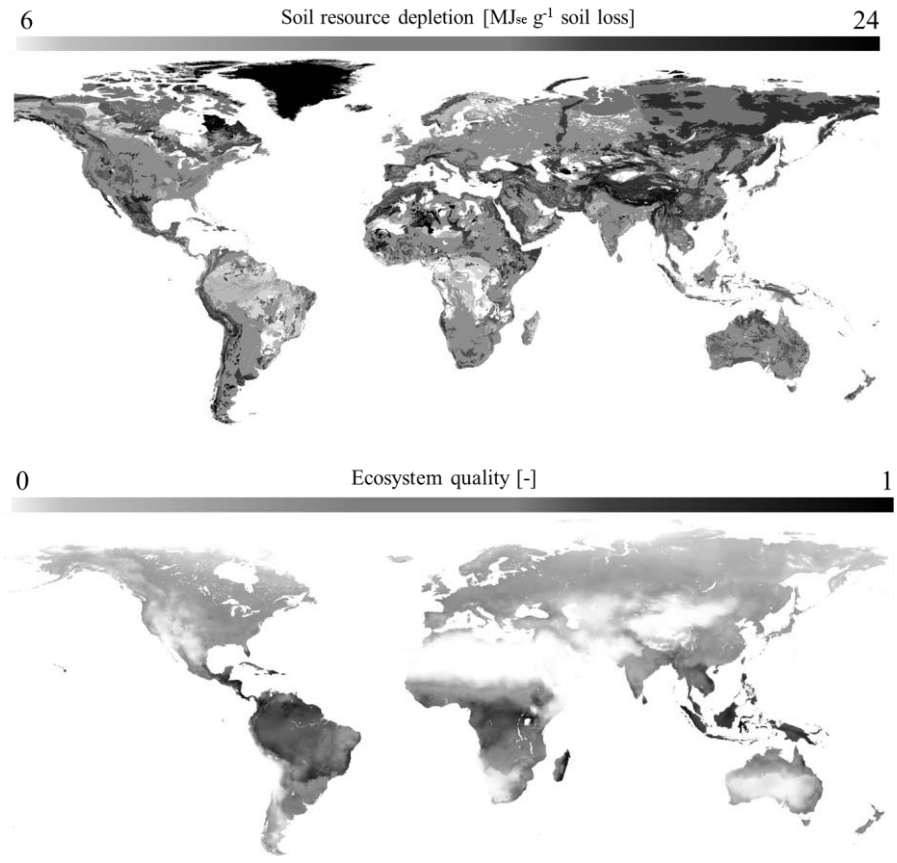


Figure 5: CFs for soil resource depletion and ecosystem quality taken from Nuñez et al. (2013). CFs derive from the model calculations.

Alvarenga et al. (2013)

This study, together with Alvarenga et al. (2015) (see just below), is one of the two models on ecosystem thermodynamics pre-selected for evaluation. The model incorporates impacts of land use interventions on the *natural resources* and *natural environment* AoPs. It computes exergy – the energy available to be used in the system – in a distinct way for natural land and for human-made land: while the exergy of the biomass extracted is calculated for natural land use types, the exergy associated to NPP is used for human-made land.

The study provides CFs at world level for a set of (twenty-seven) land use flows adapted to the Ecoinvent classification (Figure 6), and CFs at country level, which are not associated to land use flows. Both CFs typologies cover only land occupation.

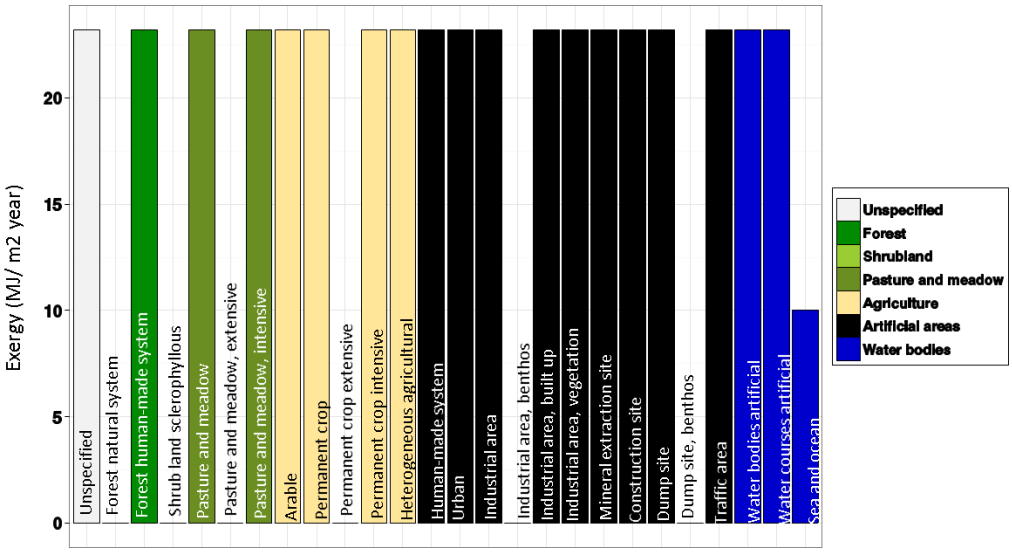


Figure 6: World CFs for land *occupation* by land use type calculated by Alvarenga et al. (2013). CFs values are very similar for most land use flows.

Alvarenga et al. (2015)

This study focuses on the Human Appropriation of Primary Production (HANPP), which is the amount of NPP that is not available for nature due to human land use. It reflects impacts of land use interventions on the *natural resources* and *natural environment* AoPs.

The study provides CFs at world level first for a few land use types at high hierarchical level (Figure 7) and then for specific crops (Figure 8).

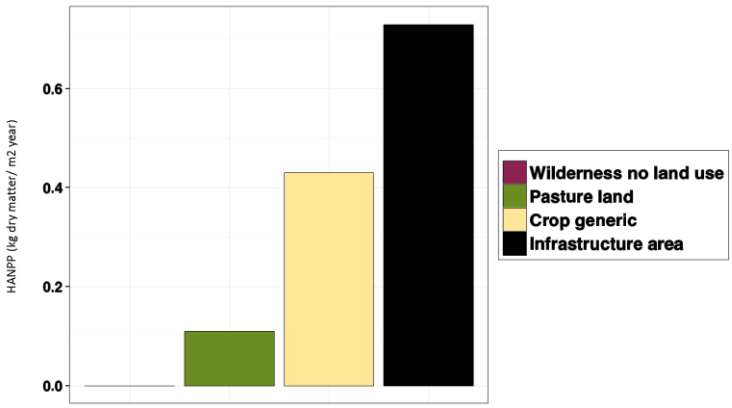


Figure 7: World CFs for land *occupation* by land use type calculated by Alvarenga et al. (2015). Land occupied by infrastructure shows the highest CFs values while wild areas reflect no impact since, due to the lack of data in HANPP from the forestry management at global scale, the authors could not generate consistent actual HANPP CFs for this land use type.

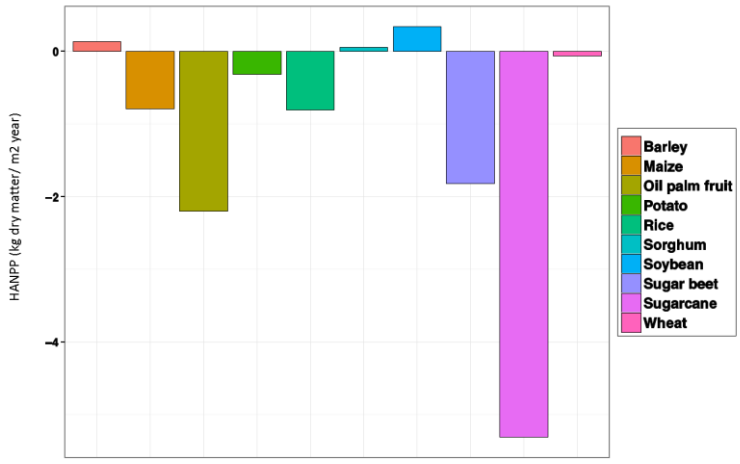


Figure 8: World CFs for land *occupation* by crop typology calculated by Alvarenga et al. (2015). The cultivation of sugarcane shows the lowest CFs value, followed by oil palm fruit and sugar beet. Conversely, soybean cultivation shows the highest value.

Gardi et al. (2013)

This study, together with Burkhard et al. (2012), is one of the two non-LCA models pre-selected for evaluation. The model makes use of spatial datasets to obtain a composite indicator on pressures to soil biodiversity. This composite indicator is a weighted index based on land use-related data (agriculture intensity, land use change), threats to soil (compaction, erosion, contamination, SOC loss), and threats to biodiversity (invasive species). The index may serve to derive biodiversity indicators, therefore covering impacts on the *natural environment* AoP.

CFs as such are not provided by the model, which generates a spatial output of the composite indicator on pressures to soil biodiversity for some European countries (Figure 9).

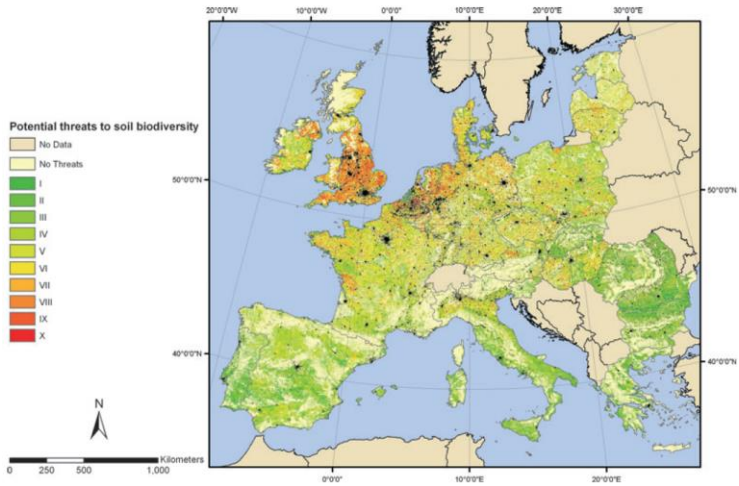


Figure 9: Spatial output of the study taken from Gardi et al. (2013), available for some European countries.

Burkhard et al. (2012)

This study calculates indicators of ecosystem integrity and ecosystem services −provisioning, regulating and cultural− directly associated to land use flows. Based on expert judgment for several case studies, the study scores the *capacity* of the different

land us flows to support a set of ecological integrity aspects and to supply a set of ecosystem services (Figure 10). The study scores as well the *demand* of the full set of indicators associated to each land use flow as well as the *budget* (i.e. the difference between capacity and demand). Among the ecosystem services indicators, the model includes soil functions (e.g. erosion regulation, water purification), and endpoint indicators (e. g. water provision). Among the ecological integrity aspects, indicators are to be found also on soil functions (SOC storage), and endpoint (biodiversity, exergy capture). Since the score obtained by each indicator is independent from the remaining indicators, the potential links between indicators that could be interpreted as consecutive in the cause-effect chain are not made explicit in this study.

Since the model was not planned put for an LCA context, no CFs as such are provided.

	Ecological Integrity								**Regulating services**									**Provisioning services**											**Cultural services**	
	Abiotic heterogeneity	Biodiversity	Biotic waterflows	Metabolic efficiency	Exergy Capture (Radiation)	Reduction of Nutrient loss	Storage capacity (SOM)		Local climate regulation	Global climate regulation	Flood protection	Groundwater recharge	Air Quality Regulation	Erosion Regulation	Nutrient regulation	Water purification	Pollination	Crops	Livestock	Fodder	Capture Fisheries	Acquaculture	Wild Foods	Timber	Wood Fuel	Energy	Biochemicals / Medicine	Freshwater	Recreation& Aesthetic Values	Intrinsic Value of Biodiversity
Continuous urban fabric	0	0	0	0	0	0	0		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Discontinuous urban	1	1	1	1	1	1	1		0	0	0	0	0	0	0	0	0	1	0	1	0	0	1	0	0	1	0	0	0	0
Industrial or commercial	1	1	0	0	0	0	0		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Road and rail networks	2	2	0	0	0	0	0		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Port areas	1	1	0	0	0	0	0		0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Airports	1	1	1	1	1	2	0		0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Mineral extraction sites	2	2	0	0	0	0	0		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0
Dump sites	2	1	0	0	0	0	5		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Construction sites	2	1	0	0	0	0	0		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Green urban areas	3	3	2	1	4	3	2		2	1	0	2	1	2	1	1	1	0	0	0	0	0	1	0	1	0	0	0	3	0
Sport and leasure facilities	2	2	2	1	4	3	2		1	1	0	2	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	5	0
Non-irrigated arable land	3	2	3	4	5	1	4		2	1	1	1	0	0	0	0	0	5	5	5	0	0	0	0	0	2	1	0	1	0
Permanently irrigated	3	2	5	2	5	1	3		3	1	1	0	0	0	0	0	0	5	5	2	0	0	0	0	0	1	1	0	1	0
Ricefields	3	2	5	1	5	1	3		2	0	0	2	0	0	0	0	0	5	0	2	0	0	0	0	0	0	0	0	1	0
Vineyards	3	2	3	1	3	0	2		1	1	0	1	0	0	0	0	0	4	0	0	0	0	0	0	1	1	0	0	5	0
Fruit trees and berries	4	3	4	2	3	2	3		2	2	2	2	2	2	1	1	5	5	0	0	0	0	0	4	4	1	0	0	5	0
Olive groves	3	2	3	2	3	1	3		1	1	0	1	1	1	1	1	0	4	0	0	0	0	0	4	4	1	0	0	5	0
Pastures	2	2	4	5	5	2	4		1	1	1	1	0	4	0	0	0	0	5	5	0	0	0	0	0	1	0	0	3	0
Annual and permanent	2	2	3	2	4	2	3		2	1	1	1	1	1	0	0	0	5	5	5	0	0	0	0	0	1	1	0	1	0
Complex cultivation	4	3	3	2	4	1	3		2	1	1	1	0	0	0	0	0	4	0	3	0	0	0	0	0	1	2	0	2	0
Agriculture& natural	3	3	3	2	3	2	3		3	2	1	2	1	3	0	1	0	3	3	2	0	0	3	3	3	2	1	0	2	3
Agro-forestry areas	4	4	4	3	4	4	4		2	1	1	1	1	2	1	1	3	3	3	2	0	0	0	3	3	2	0	0	3	0
Broad-leaved forest	3	4	5	4	5	5	5		5	4	3	2	5	5	5	5	5	0	0	1	0	0	5	5	5	1	5	0	5	5
Coniferous forest	3	4	4	4	5	5	5		5	4	3	2	5	5	5	5	5	0	0	1	0	0	5	5	5	1	5	0	5	5
Mixed forest	3	5	5	4	5	5	5		5	4	3	2	5	5	5	5	5	0	0	1	0	0	5	5	5	1	5	0	5	5
Natural grassland	3	5	4	4	4	5	5		2	3	1	1	0	5	5	5	0	0	3	0	0	0	2	0	0	0	0	0	3	3
Moors and heathland	3	4	4	5	4	5	5		4	3	2	2	0	0	3	4	2	0	2	0	0	0	1	0	2	2	0	0	5	5
Sclerophyllous vegetation	3	4	2	3	3	4	2		2	1	1	1	0	0	0	0	2	0	2	0	0	0	1	0	2	0	3	0	2	4
Transitional woodland	3	4	2	3	3	4	2		1	0	0	0	0	0	0	0	2	0	2	0	0	0	1	0	2	1	0	0	2	2
Beaches, dunes and sand	3	3	1	1	1	0	1		0	0	5	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	5	2
Bare rock	3	3	0	0	0	0	0		0	0	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	4	0
Sparsely vegetated areas	2	3	1	0	1	1	1		1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Burnt areas	2	1	0	0	0	0	3		1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glaciers and perpetual	2	1	0	0	0	0	0		3	3	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	5	0
Inland marshes	3	2	4	4	4	3	5		2	2	4	2	0	0	4	0	0	0	2	5	0	0	0	0	0	0	0	0	0	0
Peatbogs	3	4	4	4	4	5	5		4	5	3	3	0	0	3	4	2	0	0	0	0	0	0	0	2	0	0	0	4	4
Salt marshes	2	3	4	3	3	3	5		1	0	5	0	0	0	2	0	0	0	2	0	0	0	0	0	0	0	0	0	3	0
Salines	1	1	0	0	0	0	0		2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Intertidal flats	2	3	0	2	1	4	1		1	0	5	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	4	0
Water courses	4	4	0	3	3	3	1		1	0	2	1	0	0	3	3	0	0	0	0	3	0	4	0	0	3	0	5	5	5
Water bodies	4	4	0	4	4	3	4		2	1	1	2	0	0	1	0	0	0	0	0	3	0	4	0	0	0	0	5	5	4
Coastal lagoons	4	4	0	5	5	3	4		1	0	4	0	0	0	0	0	0	0	0	0	4	5	4	0	0	1	0	0	5	4
Estuaries	3	3	0	5	5	3	2		0	0	3	0	0	0	3	3	0	0	0	0	5	5	4	0	0	2	0	0	4	3
Sea and ocean	2	2	0	3	3	4	1		3	5	0	0	0	0	5	0	0	0	0	1	5	5	0	0	0	3	0	0	4	2

Figure 10: Results matrix of the assessment of the capacity of land use flows to support ecological integrity aspects and supply ecosystem services taken from Burkhard et al. (2012).
Results are based on a set of case studies and expert judgement. Similar matrices are produced for *demand* and *budget*. 0= no relevant capacity; 1= low relevant capacity; 2= relevant capacity; 3=medium relevant capacity; 4= high relevant capacity; 5= very high relevant capacity.

3.2. Additional models for comparison

In addition to the evaluated models, other reference land use-related models are described here, and their CF's reported in supplementary material (SM3). The inclusion of these models allows a more complete comparison of the ability of the evaluated models to grasps land use-related impacts. They are the following:

- The model currently recommended by the ILCD Handbook (Milà i Canals et al., 2007a; Milà i Canals et al., 2007c for the CFs)
- Cao et al. (2015), which makes a further refinement of the CFs developed by the Saad et al. (2013) model, described in section 3.1. Since this study does not provide world values, world CFs displayed below correspond to this model as it has been processed for Impact World + (Bulle et al., 2013).
- The model by de Baan et al. (2013), as representation of endpoint models, and which accounts for the impact on land interventions on biodiversity. World CFs displayed below correspond to this model as it is incorporated in Impact World + (Bulle et al., 2013)

Mila i Canals et al. (2007)

The model by Mila i Canals et al. (2007a) is the current ILCD recommendation, to be applied with caution, to estimate the impact of land use at midpoint level (EC-JRC, 2011). The model uses of soil organic matter (SOM) as an indicator of the life support function of soils. It estimates SOM losses associated to land occupation and land transformation, where a positive CF value means the land intervention leads to a SOM loss.

CFs are provided in Milà i Canals et al. (2007c) based on Ecoinvent land use flows, which were further adapted to the ILCD inventory flows (Figure 11 for occupation and Figure 12 for transformation). Although CFs values come for a United Kingdom case-study, CFs were considered for the global application of the model.

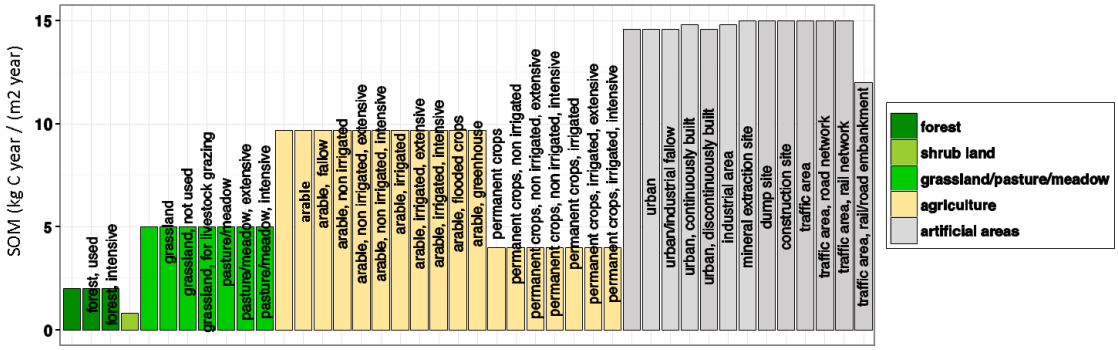


Figure 11: World CFs for land *occupation* by land use type calculated by Mila i Canals (2007a, 2007c), adaptation to ILCD flows.
Artificial areas show the highest CFs values, followed by some agricultural land use typologies.

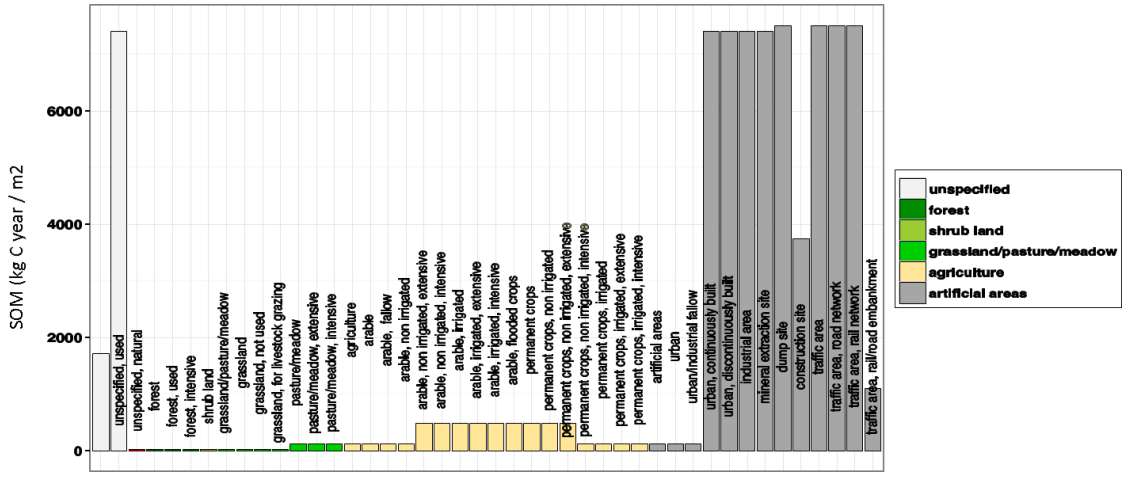


Figure 12: World CFs for land *transformation* by land use type calculated by Mila i Canals (2007a, 2007c) after adaptation to ILCD.

The impact pattern is similar to the one observed for land occupation, but with more remarkable differences between the land use typologies showing the highest CFs values.

Impact World+

Impact World+ (Bulle et al., 2013) is a LCA methods for the assessment of impacts on both ecosystem quality and on natural resource provision, including a varied set of indicators. Among those, Impact World+ considers land use impacts on biodiversity (Figure 11), providing CFs for land occupation at world scale by major land use type, based on the model by de Baan et al. (2013).

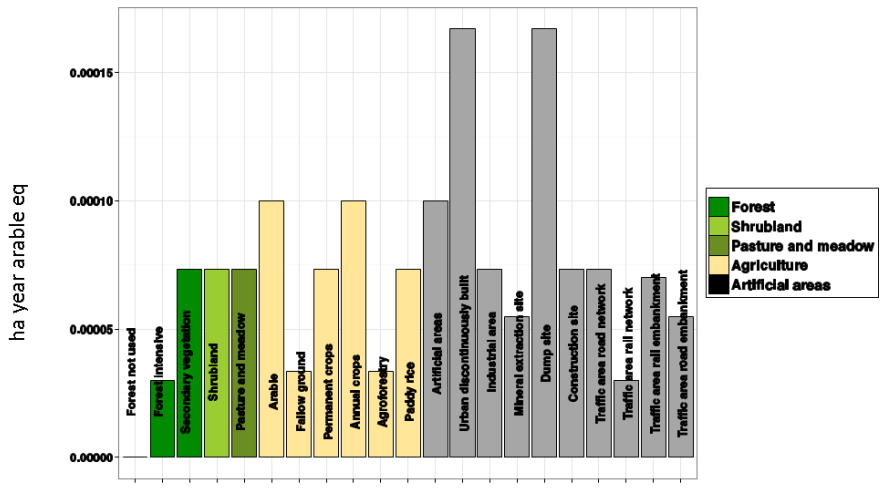


Figure 13: World CFs for land *occupation* by land use type calculated by Impact World + (Bulle et al., 2013, based on de Baan et al., 2013).

Urban and mineral site areas show the highest CFs, followed by some types of agriculture.

Cao et al. (2015)

The work developed by Cao et al. (2015) aimed at converting biophysical impact indicators from soil ecological functions into ecosystem services expressed in economic units using economic valuation. The biophysical indicators incorporated cover the following soil ecological functions: the indicators, with some modifications in the naming, used by Saad

et al. (2013) – erosion resistance, mechanical filtration, physicochemical filtration and groundwater recharge; biotic primary production based on Brandão and Mila i Canals (2013); and climate regulation potential based on Müller-Wenk and Brandão (2010). These indicators are further used as the basis to derive the monetary value of the associated ecosystem services.

The study provides CFs at country level for the whole world bot for occupation and transformation, but not world values. World CFs have been obtained from the ongoing implementation of this model to the Impact World + methodology (Figure 14 for occupation and Figure 15 for transformation).

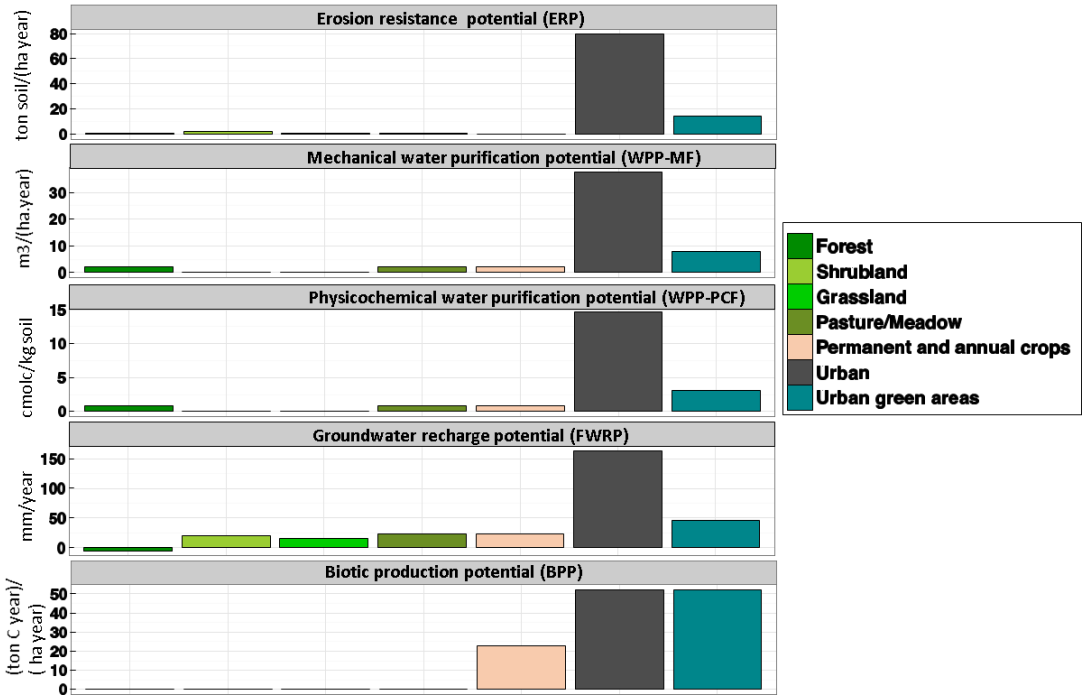


Figure 14: World CFs for land *occupation* by land use type calculated by Impact World +, based on Cao et al. (2015).
Similar impact patterns are reflected by these CFs and the ones in the original study by Saad et al. (2013), except for erosion resistance, which shows lower CFs for pasture/meadow and permanent annual crops.

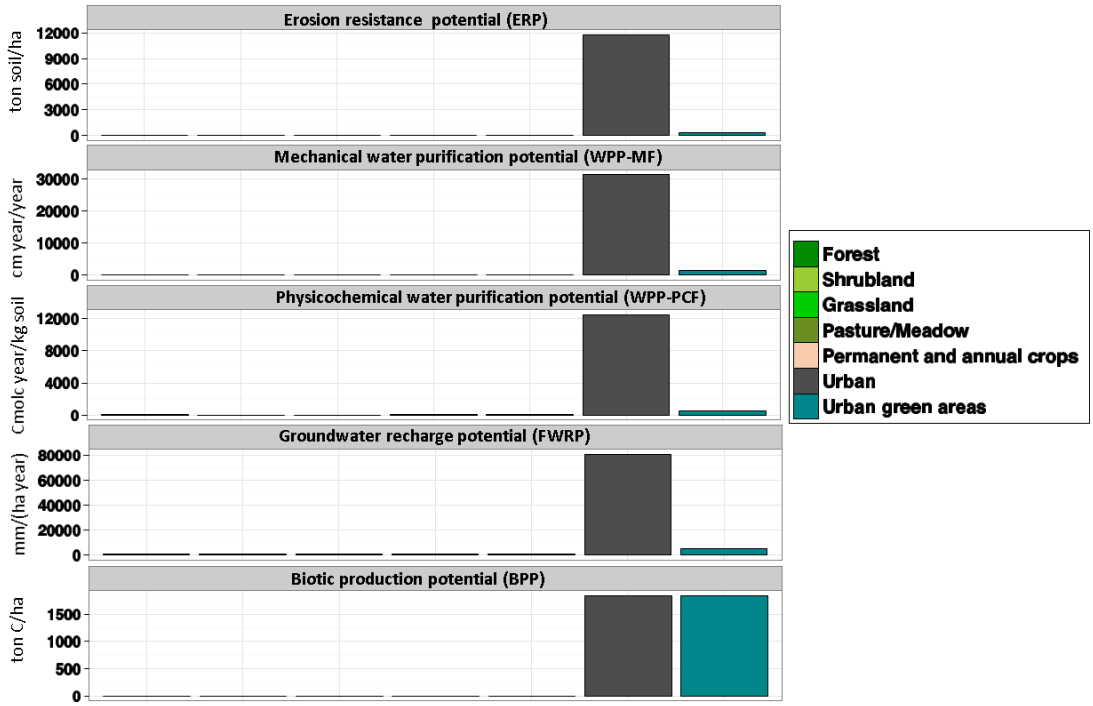


Figure 15: World CFs for land transformation by land use type calculated by Impact World +, based on Cao et al. (2015).
As it happens for land *occupation*, similar impact patterns are reflected by these CFs and the ones in the original study by Saad et al. (2013), except for erosion resistance, which shows lower CFs for pasture/meadow and permanent annual crops.

3.3. Revised land use impact pathway

The impact pathway proposed here (Figure 16) 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 other, to the soil's capacity to supply nutrients to plants (soil fertility), 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.

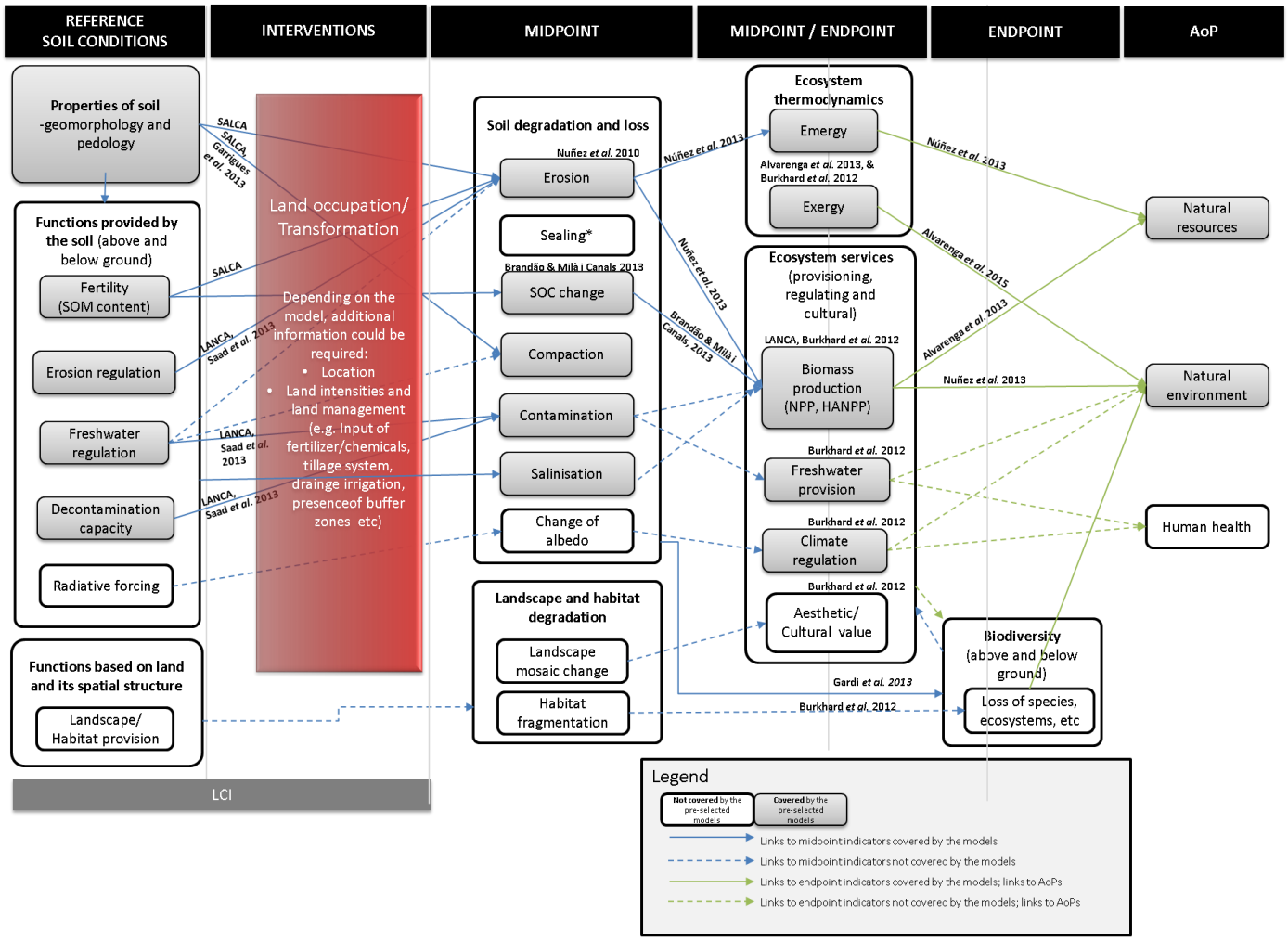


Figure 16: Revised impact pathway for land use.
The positioning of the pre-selected models is displayed as: Brandão and Milà i Canals, 2013; LANCA (Beck et al., 2010; Bos et al., 2016); Saad et al., 2013; SALCA (Oberholzer et al., 2012); Núñez et al., 2010; Garrigues et al., 2013; Núñez et al., 2013; 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.

3.4. Evaluation results

In this section the results of the models' evaluation are first presented by model (section 3.3.1) and then by set of criteria (section 3.3.2), showing how current models satisfy the need of scientifically sound and applicable impact assessment models. A summary of the evaluation scoring results is given in Table 5, while the complete filled-in evaluation form is provided in the supplementary material (SM2).

3.4.1. Evaluation results by model

Brandão and Milà i Canals (2013)

This model obtained a good evaluation in terms of impact characterisation 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; it was also challenging to fully track the sources of its input data. As for applicability, the main limitations were: i) land use flows differ from the ILCD classification, ii) low spatial resolution of the CFs, not available at country scale. Finally, the model lacks endorsement by an authoritative body.

LANCA

LANCA showed good results in terms of environmental relevance and applicability. The indicators of the model were found relevant to the ecosystem services mid-/endpoints, as well as they were placed close to inventory flows. They also covered both the *natural environment* and *natural resources* AoPs, and has a full global geographic coverage.

Aspects contributing to the good evaluation results for environmental performance were the capacity of the model to compute the impact of both land occupation and transformation, and to differentiate between extensive and intensive land uses – although the original application of the model computes the indicators referring to production units rather than land use interventions. This makes possible the use of the model at site-specific level but also at world and country level.

The model scored well in terms of robustness and uncertainty, although it shows room for improvement in terms of being up-to-date – since some of the underlying models come from the 80's and further reviews of them are not mentioned in the model documentation.

Tests of LANCA applicability have been undergone. Moreover, the model is already connected to production processes in some LCA software, although based on the application that derives indicators from production units. Yet, the latest set of CFs (Bos et al., 2016) has not yet been tested. Uncertainty figures are not available for LANCA. While the model counts on good and accessible documentation, the model itself was not available due to its private ownership. Also, access to the site-specific input is limited, which also constrains the applicability of the model. Finally, LANCA's stakeholders' acceptance was relatively low, mostly due to the complexity of the model output and the lack of underlying academic or authoritative bodies.

Saad et al. (2013)

Since Saad et al. (2013) is an application of LANCA, only the evaluation results that differ from LANCA are referred to here. The scope of Saad et al. (2013) resulted less complete than LANCA's since it does not include the biotic production calculation, which would be associated to the *natural resources* AoP.

The impact characterisation by this model is limited to only a few land use flows at high hierarchical level (e.g. shrubland, forest), and the model has not been tested as LANCA neither it is implemented in LCA software. In fact, the authors state in the study that the model was limited due to the coarse scale of application. However, Saad et al. (2013)

analyse the spatial variability of the CFs, which was already approached in a previous application of the model for Canada (Saad et al., 2011).

SALCA-SQ

While this model covers a very detailed set of soil properties and indicators, the main limitation of this model regarding completeness of scope was its foundation on site-specific data on Switzerland and the coverage of only agricultural land use flows.

The model's capacity to distinguish differential impacts on soil properties, based on extensive input data, which includes land use and land practices among other variables, is high. However, the discriminating power of the model is limited since it is based on a qualitative scale (scored as --, - , 0, +, ++). Also, guidance will be needed to assess the relevance of each of the multiple indicators that the models is able to calculate on the total impact on soil.

The scientific robustness of this model was judged to be acceptable as it was peer reviewed and scientifically validated by a case study. Yet, no estimates of uncertainty for the midpoint indicators were included, while the authors mention that not all observed impacts are consistent with model results. Documentation of the model is accessible and transparent.

Unlike many of the models, the general principles of SALCA-SQ are supported by an academic authority and a governmental body. One aspect of the stakeholder acceptance category in which this model did not score well was in neutrality, as the impact model covers only land use by agriculture.

Nuñez et al. (2010)

The model developed by Nuñez et al. (2010), which proposes a desertification index for each ecoregion in the world, obtained generally good results in terms of completeness. It can be placed as midpoint indicator in the cause-effect chain, with influence in the three AoPs, although further work would be needed to better specify these links.

In terms of environmental relevance, the model was found limited since different land use types or land practices regimes are not addressed by the model, which rather provides a CF for each ecoregion – without discriminating different land use types.

Documentation is transparent, being all the information required for the desertification impact assessment generally available. Conversely, neither the model as such nor CFs associated to land use flow inventory data were available, which does not allow for the application of the model.

Garrigues et al. (2013)

This model presents significant limitations in terms of scope and applicability. First, the model is mainly designed for agricultural production systems and does not include other land use/cover types. Second, the model requires 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. Moreover, this study provides only a limited set of CFs from site-specific applications and does not follow a land use classification compatible with Koellner et al. (2013a) and/or ILCD. Derived from all these observations, the model cannot be globally applicable. Thus, although modeling assumptions and value choices are well documented, the same is not true for the CFs. These facts also contribute to the low level of stakeholder's acceptance of the model.

Nevertheless, the requirement of local and regional data gives the model a higher spatial resolution, resulting in an improved accuracy in the prediction of potential impacts, and a better up-to-date profile.

Nuñez et al. (2013)

This model addresses two relevant impact pathways leading to the AoPs *natural environment* and *natural resources*. The model, which includes net primary production (NPP) depletion and soil loss in terms of emergy, is globally applicable and spatially explicit.

The model does not include an uncertainty assessment. However, the authors pointed out that the main uncertainty source comes from the assumption of linearity between SOC and NPP loss – several studies show the uncertainty of the link between soil erosion and biomass production. Moreover, the authors made clear that the simplifications performed to convert the great variability of soils to units of NPP or emergy would require further refinement. Model documentation is easily accessible and transparent.

As for applicability, the model is based, among other, on land management and land use variables, 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. Finally, although erosion equations are commonly accepted, they are neither included in LCA software nor their conversion to damage units has been endorsed by any scientific or policy authority. This partly explains the low score recorded for stakeholders' acceptance.

Alvarenga et al. (2013)

The model by Alvarenga et al. (2013), approaching the impact of land use on the potential net primary production expressed in exergy units, showed good performance in terms of completeness of scope, covering the *natural resources* AoP (directly, as it deals with land as a resource), and (indirectly) the *natural environment* AoP.

This model has limited environmental relevance mostly because it gives a poor impact characterization: CFs values are often the same for the different land use types. Moreover, although CFs are directly linked to land use flows, the number of land use flows covered and specifically the coverage of ILCD land use inventory flows are very limited. 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 from the aggregation process at country level rather than as an error measure that involves the uncertainty of the original data, whose quality is not assessed. The model is well documented. Finally, the model ranks low on stakeholder acceptance given the lack of institutional support.

Alvarenga et al. (2015)

This model, which uses HANPP as proxy of aspects related to soil quality, biotic production and biodiversity, ranked high in terms of completeness of scope. It also 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 characterisation.

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 readily usable in LCA and matches with some ILCD land use flows.

Finally, the HANPP concept is potentially harder to explain to non-experts than alternative land use indicators, which explains why it ranks lower in stakeholder acceptance (besides lacking institutional support).

Gardi et al. (2013)

This non-LCA model performed well on the consistency between those indicators that may be used as midpoints and endpoints, as well as on the relation between inventory data and midpoints. Although the model has high spatial resolution (1x1 km grid cells), its geographic coverage is limited to Europe. The model computes the pressures on soil biodiversity 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 for the model.

Burkhard et al. (2012)

The set of indicators computed by the Burkhard et al. (2012) model has an extensive coverage of all AoPs. While some of the indicators adopted in this model could be considered midpoint indicators, others would be closer to endpoints. However, no links between midpoint and endpoints are established by the model since all indicators are directly and solely derived from land use types.

This model scored well in environmental relevance, due to the inclusion of this comprehensive set of indicators, which covers ecosystem services, biodiversity, climate change, cultural value, etc. However, this was offset slightly by the lack of characterisation testing and the lack of guidance for a possible aggregation of the information contained in the different indicators.

The model results are not mathematically reproducible since the model 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.

Burkhard et al. (2012) is the product of individual research and remains neutral across industries, products, and/ or processes and produces easily understandable results. Currently, there is no endorsement by authoritative entities for the use of this model, but the characterisation model results could be relevant for current policy, since they target many policy areas.

3.4.2. Evaluation results by 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 models, explicit links were found only to the AoPs *natural resources* and *natural*

environment. As for geographic coverage (see Table 4), models generally allow for the global application of the model, except for SALCA-SQ −site-specific−, Gardi et al., (2013) −only for Europe−, and Garrigues et al. (2013) and Burkhard et al., (2012) – both based on three local case studies.

Environmental relevance

The models proposed by Brandão and Milà i Canals (2013) and Gardi et al. (2013) allowed for the most relevant characterisation of the impact of land interventions, also under different management practices schemes. The characterisation 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). For the models providing CFs also for land transformation, similar impact patterns were found as compared to the impact of land occupation.

Overall, artificial land uses showed the strongest impact for all models and indicators except for the calculation of the impact of land interventions on the soil erosion resistance by LANCA, for which bare areas pose the strongest impact. Interestingly, the pattern of the impact by land intervention differs between the model based on SOC calculations – Brandão and Milà i Canals (2013) – and the LANCA module for biotic production. Also, although the scope of multi-indicators models appears as superior, in some cases the information provided by the different indicators was found redundant.

Finally, the temporal resolution of the models was generally annual, being the spatial resolution overall higher than national.

Scientific robustness & 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, models were very limited overall, except for Gardi et al. (2013).

Documentation, transparency & 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 practice, while the remaining models 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 characterisation model, and the fact that the model itself was not available in an operational format.

Table 5: Results of the models' evaluation.

The scores obtained by each model for each criterion is reported, ranging between A and E – which generally mean, respectively, the best and worst possible result. 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	Saad et al. 2013	SALCA	Núñez et al. 2010	Garrigues et al. 2012	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 characterisation 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 characterisation	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	• Characterisation 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 characterisation model documentation	A	A	A	A	A	C	A	A	A	C	A
	• Completeness of the characterisation 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 characterisation factors	A	A	A	E	A	E	A	A	A	E	C
	• Accessibility of the characterisation model	A	E	E	C	E	D	E	C	C	E	E
	• Modelling 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 characterisation 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 characterisation factors	D	D	D	E	D	D	D	D	D	D	A
	• Spatial resolution of characterisation 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

Applicability

LANCA in its latest version (Bos et al. 2016) stood out in terms of applicability, with CFs available both at world level and at country scale. The application of LANCA by Saad et al. (2013) also stood out in these terms, although allowing only for the characterisation of a limited number of coarse land use inventory flows.

Overall, the assessed models have the required LCI flows available, although not always complete and not corresponding to the recommended classification. 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 applicability. The models by Núñez et al., (2010, 2013) and Garrigues et al. (2013) were based on inventory flows that 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 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.

The usability of the CFs was the main issue in terms of applicability. Although CFs for a global application were available for the majority of the models, not all were associated to land use inventory flows neither to a geographic scale (e.g. country) that facilitates implementation in LCA software. 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. Apart from that, none of the evaluated models provided normalization factors.

The non-LCA models Gardi et al. (2013) and Burkhard et al. (2012) did not provide strictly speaking CFs. However, the output by Burkhard et al. (2012) could be relatively easily assimilated as such.

Stakeholders' acceptance

Models were overall limited in terms of stakeholder acceptance, mostly due to missing authoritative/academic bodies. Best results, with a remarkable difference, were for Gardi et al. (2013) and the lowest for the model by Saad et al. (2013).

Neutrality was challenged for some models mostly due to the limited coverage of flows by the impact assessment, e.g. models only addressing agricultural activities/land use types –e.g. SALCA-SQ, and Garrigues et al. (2013).

Gardi et al. (2013) was found the most policy-relevant model, having indicators already present in current reference European soil policy documents (The European Atlas of Soil Biodiversity- Jeffery et al., 2010).

4. Discussion and outlook

The models reviewed in this study were 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 characterisation 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 of land interventions 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 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 and 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, and 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 of land interventions 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.

Additional limitations in terms of applicability were also found: i) CFs for land transformation were missing for most models, and ii) guidance for the calculation of normalization factors was totally absent from all models.

4.3. Single indicator vs. multi-indicator models

The need for 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 et al., 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, the comparison of multi-indicator models' CFs shows that the information given by some of these indicators could be redundant – e.g. Helin et al. (2014) found a high correlation among some indicators within the set proposed in Saad et al. (2013). This points out several research needs, including: i) statistical analyses of the redundancy of CF values in multi-indicator models; ii) 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?". Furthermore, 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 first provide non-redundant information, and then either provide guidance on how to assess the relevance of each of the multiple indicators or propose an aggregation algorithm.

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 full environmental impact on the soil.

4.4. 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 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 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 as 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.

4.5. 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 reflect the differences amongst land use intensities and 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 CFs 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.

In this study, the need for a more comprehensive land use classification was identified, fact that was already previously pointed out by Allacker et al. (2014). This is due to the fact that 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 LCA 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 will be important to assess the data limitations encountered both by LCA practitioners and by model developers.

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 given the needs of the land use models. To this regard, building the inventory based on land use flows is questioned by Helin et al. (2014), who 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.

Although we can agree that the land interventions inventory flows do not provide the full picture of the impact, it is important to bear in mind that making inventories very complicated (including e.g. data on soil conditions, climate, or location) could hampered the LCA practice (mainly when the location of activities is unknown). We are facing a complex system with a lot of interactions and specific features for which is difficult to balance between comprehensive and feasible data. Therefore, it is clear that it is necessary an agreement of the scientific community on the detail of assessment and what needs to be answered by an LCA study. A starting point to address this question could be to build a limited number of archetypes covering the questions of what (e.g. forest, fruit crops), where (e.g. ecoregions, countries), which extension, and how (intensiveness) land is occupied, but of course that must be done in close cooperation with models developers, in order to have the appropriate CFs for the different archetypes.

4.7. Land use impact pathway and its link with climate change and resource availability

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 soils also play 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. Characterisation factors limitations and uncertainties

Our evaluation was based also in a preliminary analysis of the CFs. A further quantitative assessment of the models' CFs 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 CFs spatial variability, and the relative relevance of the impacts resulting from land occupation and transformation. 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.

5. Supplementary material

This report is accompanied by supplementary material (SM) which could be downloaded from http://eplca.jrc.ec.europa.eu/uploads/supplementary-material-land-use.xlsx

The SM includes: SM1- criteria and sub-criteria for the models' evaluation; SM2- the filled-in models' evaluations; SM3- compilation of CFs at world level provided by the models, adapted to the ILCD elementary flows (when they were available with a different land use classification).

6. References

Allacker, K., Souza, D.M. de, Sala, S., 2014. Land use impact assessment in the construction sector: an analysis of LCIA models and case study application. Int. J. Life Cycle Assess. 19, 1799–1809.

Alvarenga, R.F., Dewulf, J., Van Langenhove, H., Huijbregts, M.J., 2013. Exergy-based accounting for land as a natural resource in life cycle assessment. Int. J. Life Cycle Assess. 18, 939–947.

Alvarenga, R.F., Erb, K.-H., Haberl, H., Soares, S., van Zelm, R., Dewulf, J., 2015. Global land use impacts on biomass production—a spatial-differentiated resource-related life cycle impact assessment method. Int. J. Life Cycle Assess. 20, 440–450.

Baitz, M., 2002. Die Bedeutung der funktionsbasierten Charakterisierung von Flächen-Inanspruchnahmen in industriellen Prozesskettenanalysen: Ein Beitrag zur ganzheitlichen Bilanzierung. PhD dissertation. Institut für Kunststoffprüfung und Kunststoffkunde, Universität Stuttgart. Shaker Verlag.

Beck, T., Bos, U., Wittstock, B., Baitz, M., Fischer, M., Sedlbauer, K., 2010. LANCA® Land Use Indicator Value Calculation in Life Cycle Assessment – Method Report', Fraunhofer Institute for Building Physics.

Bos, U., Horn, R., Beck, T., Lindner, J.P., Fischer, M., 2016. LANCA® - Characterisation Factors for Life Cycle Impact Assessment, Version 2.0, ISBN: 978-3-8396-0953-8, Fraunhofer Verlag, Stuttgart.

Brandão, M., Mila i Canals, L., 2013. Global characterisation factors to assess land use impacts on biotic production. Int. J. Life Cycle Assess. 18 (6), 1243-1252.

Bulle, C., Margni, M., Jolliet, O., Rosenbaum, R., Humbert, S., 2013. IMPACT World+. from http://www.impactworldplus.org/en/.

Burkhard, B., Kroll, F., Nedkov, S., Müller, F., 2012. Mapping ecosystem service supply, demand and budgets. Ecol. Indic. 21, 17–29.

Cao, V., Margni, M., Favis, B.D., Deschênes, L., 2015. Aggregated indicator to assess land use impacts in life cycle assessment (LCA) based on the economic value of ecosystem services. J. Clean. Prod. 94, 56–66.

COM 2006 231 final. Thematic Strategy for Soil Protection. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions.

Cowell, S., Clift, R., 1997. Impact assessment for LCAs involving agricultural production. Int. J. Life Cycle Assess. 2, 99–103.

Cowell, S.J., Clift, R., 2000. A methodology for assessing soil quantity and quality in life cycle assessment. J. Clean. Prod. 8, 321–331.

Curran, M.P., Souza, D.M., Antòn, A., Teixeira, R.F.M., Michelsen, O., Vidal-Legaz, B., Sala, S., Milà i Canals, L. How well does LCA model land use impacts on biodiversity? —A comparison with approaches from ecology and conservation. Environ. Sci. Technol. *In press*.

De Baan, L., Alkemade, R., Koellner, T., 2013. Land use impacts on biodiversity in LCA: a global approach. Int. J. Life Cycle Assess. 18, 1216-1230.Del Borghi, A., Gallo, M., Strazza, C., Del Borghi, M., 2014. An evaluation of environmental sustainability in the food industry through Life Cycle Assessment: the case study of tomato products supply chain. J Clean Prod, 78, 121-130.

Dewulf, J., Bösch, M.E., Meester, B.D., Vorst, G.V. der, Langenhove, H.V., Hellweg, S., Huijbregts, M.A.J., 2007. Cumulative Exergy Extraction from the Natural Environment (CEENE): a comprehensive Life Cycle Impact Assessment method for resource accounting. Environ. Sci. Technol. 41, 8477–8483.

Doran, J.W., 2002. Soil health and global sustainability: translating science into practice. Agric. Ecosyst. Environ. 88, 119–127.

Doran, J. W., Parkin, T. B., 1996. Quantitative indicators of soil quality: a minimum data set. In Doran, J.W. and Jones, A.J. (eds). Methods for Assessing Soil Quality. Madison, Wisconsin, USA.

EC (2013). Commission Recommendation of 9 April 2013 on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations. ANNEX II: Product Environmental Footprint (PEF) Guide. European Commission

EC-JRC (European Commission -Joint Research Centre), 2011. Recommendations based on existing environmental impact assessment models and factors for life cycle assessment in European context. Luxembourg: Publications Office of the European Union. EUR24571EN. ISBN 978-92-79- 17451-3. Available at http://eplca.jrc.ec.europa.eu/.

EC-JRC (European Commission -Joint Research Centre), 2012. Characterisation factors of the ILCD Recommended Life Cycle Impact Assessment methods. Database and Supporting Information. First edition. EUR 25167. Luxembourg. Publications Office of the European Union; 2012. Technical note and latest characterisation factors (CFs) available at: http://eplca.jrc.ec.europa.eu/?page_id=1159.

Fehrenbach, H., Grahl, B., Giegrich, J., Busch, M., 2015. Hemeroby as an impact category indicator for the integration of land use into life cycle (impact) assessment. Int. J. Life Cycle Assess. 20(11), 1511−1527.

Gardi, C., Jeffery, S., Saltelli, A., 2013. An estimate of potential threats levels to soil biodiversity in EU. Glob. Chang. Biol. 19, 1538–1548.

Garrigues, E., Corson, M.S., Angers, D.A., van der Werf, H.M.G., Walter, C., 2012. Soil quality in Life Cycle Assessment: Towards development of an indicator. Ecol. Indic. 18, 434–442.

Garrigues, E., Corson, M., Angers, D., van der Werf, H.G., Walter, C., 2013. Development of a soil compaction indicator in life cycle assessment. Int. J. Life Cycle Assess. 18, 1316–1324.

Heijungs, R., Guinee, J., Huppes, G., 1997. Impact categories for natural resources and land use. Leiden University. Leiden, The Netherlands. 40pp.

Helin, T., Holma, A., Soimakallio, S., 2014. Is land use impact assessment in LCA applicable for forest biomass value chains? Findings from comparison of use of Scandinavian wood, agro-biomass and peat for energy. Int. J. Life Cycle Assess. 19, 770–785.

Hellweg, S., Milà i Canals, L., 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science. 344(6188), 1109-113.

Jandl, R., Rodeghiero, M., Martinez, C., Francesca Cotrufo, M., Bampa, F., van Wesemael, B., Harrison, R.B., Amaral Guerrini, I., Richter Jr, D.B., Rustad, L., Lorenz, K., Chabbi, A., Miglietta, F., 2014. Current status, uncertainty and future needs in soil organic carbon monitoring. Sci Total Environ. 468-469, 376-383.

Jeffery, S., Gardi, C., Jones, A. et al., 2010. The European Atlas of Soil Biodiversity. Publications office of the European Union, Luxembourg.

Jolliet, O., Müller-Wenk, R., Bare, J., Brent, A., Goedkoop, M., Heijungs, R., Weidema, B., et al., 2004. The LCIA midpoint-damage framework of the UNEP/ SETAC life cycle initiative. Int. J. Life Cycle Assess. 9 (6), 394e404.

Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Goedkoop, M., Margni, M., i Canals, L., Müller-Wenk, R., Weidema, B., Wittstock, B., 2013a. Principles for life cycle inventories of land use on a global scale. Int. J. Life Cycle Assess. 18, 1203–1215.

Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., i Canals, L., Saad, R., de Souza, D., Müller-Wenk, R., 2013b. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. Int. J. Life Cycle Assess. 18, 1188–1202.

Li, G., Chen, J., Sun, Z., Tan, M., 2007. Establishing a minimum dataset for soil quality assessment based on soil properties and land-use changes. Acta Ecol. Sin. 27, 2715–2724.

Liedtke, C., Baedeker, C., Kolberg, S., Lettenmeier, M., 2010. Resource intensity in global food chains: the Hot Spot Analysis. Brit Food J, 112, 1138 – 1159.

Lindeijer, E., 2000. Review of land use impact methodologies. J Clean. Prod. 8, 273–281.

Mattila, T., Seppälä, J., Nissinen, A., Mäenpää,I., 2011. Land Use Impacts of Industries and Products in the Finnish Economy: A Comparison of Three Indicators. Land Use Impacts of Bioenergy. Selected Papers from the IEA Bioenergy Task 38 Meetings in Helsinki, 2009 and Brussels, 2010 35, no. 12: 4781–87.

Mattsson, B., Cederberg, C., Blix, L., 2000. Agricultural land use in life cycle assessment (LCA): case studies of three vegetable oil crops. J. Clean. Prod. 8, 283–292.

Milà i Canals, L., Romanyà, J., Cowell, S.J., 2007a. Method for assessing impacts on life support functions (LSF) related to the use of 'fertile land' in Life Cycle Assessment (LCA). J Clean Prod. 15, 1426–1440.

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Knuchel, R.F., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007b. Key elements in a framework for land use impact assessment within LCA. Int. J. Life Cycle Assess. 12(1), 5-15.

Milà i Canals, L., 2007c. LCA methodology and modeling considerations for vegetable production and consumption. Centre for Environmental Strategy, University of Surrey.

Müller-Wenk, R., Brandão, M., 2010. Climatic impact of land use in LCA—carbon transfers between vegetation/soil and air. Int. J. Life Cycle Assess., 15, 172–182.

Núñez, M., Civit, B., Muñoz, P., Arena, A., Rieradevall, J., Antón, A., 2010. Assessing potential desertification environmental impact in life cycle assessment. Int. J. Life Cycle Assess. 15, 67–78.

Núñez, M., Antón, A., Muñoz, P., Rieradevall, J., 2013. Inclusion of soil erosion impacts in life cycle assessment on a global scale: application to energy crops in Spain. Int. J. Life Cycle Assess. 18, 755–767.

Oberholzer, H.-R., Freiermuth Knuchel, R., Weisskopf, P., Gaillard, G., 2012. A novel method for soil quality in life cycle assessment using several soil indicators. Agron. Sustainable Dev. 32, 639–649.

Payen, S., Basset-Mens, C., Follain, S., Gruenberger, O., Marlet, S., Núñez, M., Perret, S., 2014. Pass the salt please! From a review to a theoretical framework for integrating salinization impacts in food LCA. In: Schenck, R., and Huizenga, D. (eds). Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food sector. American Center for Life Cycle Assessment.

Saad, R., Margni, M., Koellner, T., Wittstock, B., Deschênes, L., 2011. Assessment of land use impacts on soil ecological functions: development of spatially differentiated characterization factors within a Canadian context. Int. J. Life Cycle Assess. 16, 198–211.

Saad, R., Koellner, T., Margni, M., 2013. Land use impacts on freshwater regulation, erosion regulation, and water purification: a spatial approach for a global scale level. Int. J. Life Cycle Assess. 18, 1253–1264.

Souza, D.M., Teixeira, R.F.M, Ostermann, O., 2015. Assessing biodiversity loss due to land use with Life Cycle Assessment: Are we there yet? Glob. Chang. Biol. 21, 32-47.

Teixeira, R.F.M., Maia de Souza, D., Curran, M.P., Antón, A., Michelsen, O., Milà i Canals, L., 2016. Towards consensus on land use impacts on biodiversity in LCA: UNEP/SETAC Life Cycle Initiative preliminary recommendations based on expert contributions. J. Clean. Prod. 112, Part 5, 4283–4287.

Vidal Legaz B, De Souza D M, Teixeira R, Anton A, Putman B, Sala S, (2016). Soil quality, properties, and functions in Life Cycle Assessment: an evaluation of models Journal of Cleaner Production, p 1-14, doi: 10.1016/j.jclepro.2016.05.077

Weidema, B.P., Lindeijer, E., 2001. Physical impacts of land use in product life cycle assessment: Final report of the EURENVIRONLCAGAPS sub-project on land use. IPL-033-01, Lyngby.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall erosion losses—a guide to conservation planning. Agricultural Handbook no. 537. Department of Agriculture, USA.

List of figures

List of tables

How to obtain EU publications

Our publications are available from EU Bookshop (http://bookshop.europa.eu),

where you can place an order with the sales agent of your choice.

The Publications Office has a worldwide network of sales agents.

You can obtain their contact details by sending a fax to (352) 29 29-42758.

Europe Direct is a service to help you find answers to your questions about the European Union

Free phone number (*): 00 800 6 7 8 9 10 11

(*) Certain mobile telephone operators do not allow access to 00 800 numbers or these calls may be billed.

JRC Mission

As the Commission's in-house science service, the Joint Research Centre's mission is to provide EU policies with independent, evidence-based scientific and technical support throughout the whole policy cycle.

Working in close cooperation with policy Directorates-General, the JRC addresses key societal challenges while stimulating innovation through developing new methods, tools and standards, and sharing its know-how with the Member States, the scientific community and international partners.

Serving society
Stimulating innovation
Supporting legislation

Publications Office

doi:10.2788/905478

ISBN 978-92-79-54023-3